# ADVANCING SUSTAINABLE WASTEWATER TREATMENT: ELUCIDATING TRADEOFFS AMONG EMERGING RESOURCE RECOVERY TECHNOLOGIES THROUGH QUANTITATIVE SUSTAINABLE DESIGN

BY

BRIAN D. SHOENER

THESIS

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Adviser:

Assistant Professor Jeremy S. Guest

# ABSTRACT

Anthropogenic activities are negatively impacting the environment through biodiversity loss, altering nutrient cycles, and increases in severe weather events. These impacts are subsequently hindering the ability of wastewater treatment plants (WWTPs) to protect human and environmental health. In addition, the field of wastewater engineering is facing several problems that must be addressed in the coming decades, such as aging infrastructure and stricter effluent discharge requirements. Wastewater treatment is currently primarily based on the cultivation of aerobic heterotrophs and though it provides a high-quality effluent, it is also energy intensive. High energy demand is costly both economically and environmentally. These problems underlie a need to re-envision WWTPs as a resource capable of nutrient and energy recovery while continuing to hold human and environmental health paramount.

In order to compare possible approaches to solving the problems facing wastewater treatment, a critical review was conducted comparing several anaerobic and phototrophic technologies to determine their potential for energy positive domestic wastewater treatment. Phototropic processes were shown to be able to produce 280-400% greater energy than anaerobic processes producing methane (on a per $m^3$ basis). However, phototrophic processes increase chemical oxygen demand (COD), so a downstream process is also necessary. Anaerobic membrane bioreactors (AnMBRs) were found to have the highest consistent COD removal (80-90%) of the anaerobic processes, but also had high energy consumption. Though they are a new technology, AnMBRs show promise for full-scale domestic wastewater treatment, but because there are many different designs available, research on the topic varies greatly. An in-depth examination of AnMBR designs was conducted utilizing quantitative sustainable design to elucidate configurations that limit economic or environmental impacts under the assumption that all designs treat wastewater to the same effluent quality. The results show that certain design decisions have a profound impact on the total net present cost and life cycle environmental impacts. Therefore, recommendations for future research are made that traverse the relative benefits and detriments of different AnMBR configurations.

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# ATTRIBUTION

Dr. Jeremy S. Guest, *Ph.D.*, Assistant Professor, Department of Civil and Environmental Engineering, University of Illinois at Urbana-Champaign, Urbana, IL 61801

Dr. Roland D. Cusick, *Ph.D.*, Assistant Professor, Department of Civil and Environmental Engineering, University of Illinois at Urbana-Champaign, Urbana, IL 61801

Ian M. Bradley, *M.S.*, Graduate Research Assistant, Department of Civil and Environmental Engineering, University of Illinois at Urbana-Champaign, Urbana, IL 61801

Dr. Guest, Dr. Cusick, and Mr. Bradley are duly credited as coauthors of the manuscript cited for Chapter 2 of this document. Dr. Guest served as the Principal Investigator for the research described in Chapters 2 and 3 of this document.

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# CHAPTER 1

# INTRODUCTION

Given that there is 97% agreement among peer-reviewed literature discussing climate change that humans have played a role in influencing global warming [1], people have come to understand that the Earth's ability to sustain a stable global environment is being strained. These anthropogenic activities are proving to have serious consequences, such as loss of biodiversity, changes in nutrient cycles, and increases in severe weather events [2,3]. In response to these complex problems, sustainability science and engineering seeks to cultivate cooperative efforts that address the dynamic (and sometimes dichotomous) interactions between society and nature [4, 5]. Within the field of wastewater engineering, sustainability is still in its infancy. However, given that these global impacts are negatively affecting wastewater treatment plants' ability to protect public health and waterways [6], a need has arisen to advance sustainable wastewater treatment in order to stave off future problems.

The field of wastewater engineering is dedicated to ensuring the continued resilience of human and environmental health, but the convergence of several critical challenges has made this goal increasingly difficult to accomplish. These include: aging infrastructure in serious need of repair [7], upgrades required in order to meet increasingly strict effluent requirements [8], rapid urban population growth [9–11], increases in nutrient (i.e., nitrogen and phosphorus) loading [12], and deteriorating freshwater resources [13]. Updating this crumbling infrastructure is expected to cost close to $300 billion over the next 20 years [7]. On top of this, the predominant approach to wastewater treatment (i.e., utilizing activated sludge) is energy intensive, accounting for approximately 3% of the electricity demand in the U.S. [14], which indirectly emits greenhouse gases when fossil fuels are used to produce electricity [15]. Taken together, these stresses underlie a need to re-envision the design and operation of wastewater treatment plants from a reaction to human-made waste to a renewable resource capable of nutrient and energy recovery [16].

As a concept, resource recovery from wastewater has existed for well over a century [17] and is typically focused on recovering water, energy, and nutrients [18]. However, the reliance on aerobic processes severely hinders energy and nutrient recovery. Alternative technologies do exist that are capable of recovering these resources more effectively, namely anaerobic and phototrophic processes [19]. Anaerobic technologies recover energy from organic carbon in the form of methane, hydrogen gas, or direct electricity [19]. Phototrophic technologies cultivate microorganisms that assimilate nutrients and can be used as bioenergy feedstock [20]. Generally speaking, these technologies have been around for many years [21, 22], but they were often used when operational experience and/or capital was limited [23, 24]. With the challenges facing wastewater engineering and the Earth as a whole, the focus of both academia and industry needs to be broadened to better understand the full potential for energy recovery and production by anaerobic and phototrophic processes.

Apart from converting organic carbon to recoverable energy, anaerobic processes possess other benefits over conventional aerobic wastewater treatment. For example, energy consumption (and therefore operational cost) is often lower because aeration is not needed and slower growth kinetics result in less sludge wasting [25, 26]. The slower kinetics also manifest in a longer startup time, difficulty recovering from shocks to the system, and longer solids residence times (SRTs), the latter of which can necessitate larger plant footprints. When treating municipal-strength wastewater, anaerobic microbes aggregate poorly, resulting in poor settling characteristics and frequent biomass loss in the effluent [27]; this sometimes results in failure to meet discharge requirements. Additionally, while these processes often produce methane, between 30-50% of the methane can remain dissolved in the effluent stream [28, 29]. Fugitive methane is a serious problem given that methane is 28 times worse than $CO_2$ as a greenhouse gas [2]. Tantamount to this, released methane is lost energy that could have been used to produce electricity.

Anaerobic membrane bioreactors (AnMBRs) are a comparatively new anaerobic technology that has the capability to overcome these limitations by combining an anaerobic treatment process with membrane filtration [30]. Including a membrane during treatment allows hydraulic retention time (HRT) to be decoupled reliably from SRT, which results in a smaller footprint, lowering capital costs [31]. Paramount to these advantages is the AnMBR's ability to provide robust, resilient solid-liquid separation resulting in a consistently high-quality effluent [19]. In an industry focused on protecting human and environmental health, this quality is a particular asset for AnMBRs. Of particular importance is removal of organic matter across the membrane. Though the mechanism by which this is accomplished is not fully known (i.e., physical separation, degradation by microorganisms on the membrane, or a combination of both), multiple studies have shown that chemical oxygen demand (COD) removal takes place within the biofoulant, further improving effluent quality over more widely-used anaerobic technologies [32–35]. However, for all the benefits this technology has, more energy is consumed during the filtration process due in large part to membrane fouling control (e.g., gas sparging) [19, 36]. This technology has great potential to be viable as a full-scale treatment plant, but for this precise reason, many designs exist [28, 32–34, 37–44]; some of these designs have competitive advantages that need to be ascertained and exploited so that research can be more focused going forward. Determining what these advantages are requires examining this technology not only for its economic costs and benefits, but also how it impacts the environment.

The example of the AnMBR underlies a need for a paradigm shift from looking at just the net present cost of a wastewater treatment plant to examining both the economic and environmental impacts throughout the life cycle of the plant. Drawing from the main tenets of sustainable design in the context of wastewater management [16], the landscape of possible approaches to solving the problem of wastewater treatment should be compared. Situations vary between design projects, so there is no blanket solution. However, by identifying paths that have serious shortcomings, future economic and environmental detriments can be avoided.

Ultimately, the goal of this thesis is to advance wastewater treatment sustainability. To that end, and in order to understand the capabilities of AnMBRs in the context of the spectrum of alternative wastewater treatment technologies, several anaerobic and phototrophic technologies were compared to explicate their potential for energy positive domestic wastewater treatment (Chapter 2). An in-depth examination of AnMBR designs was also conducted to determine its potential in terms of treatment efficacy and environmental benefit (Chapter

3). Elucidating designs that possess competitive advantages or hindering pitfalls across a wide range of inputs provides a unique opportunity to compare different configurations in an unbiased manner. In this way, the AnMBR is examined to determine its ability to overcome longstanding problems that plague traditional anaerobic processes while also upholding the primary goal of maintaining the well being of society and the environment.

# CHAPTER 2

# ENERGY POSITIVE DOMESTIC WASTEWATER TREATMENT: THE ROLES OF ANAEROBIC AND PHOTOTROPHIC TECHNOLOGIES

## 2.1 Introduction

The sanitation industry is facing a confluence of events that are straining utility budgets [6,9] and reducing their ability to provide reliable protection of public health and the aquatic environment [6]. Critical challenges include rapid and localized population growth and decay [9–11,45]; aging and deteriorating infrastructure [7]; deterioration of surface waters resulting from excess nutrient (N and P) loading [12, 46–48]; and a reliance on expensive, energy-intensive [8,14] treatment processes. These pressures are exacerbated by decreased resilience of ecosystems [49–52] and increased variability in renewable freshwater resources [13, 53, 54] resulting from climate change, with current energy-intensive approaches to wastewater treatment (consuming roughly 0.3-0.6 kWh·m$^{-3}$ of wastewater treated [55,56], or 3% of U.S. electricity demand [14]) further contributing to climate change via greenhouse gas emissions from electricity production [15, 57]. Ultimately, these stressors have intensified the need to address the water-energy nexus in wastewater management. Given that upgrades to U.S. infrastructure are expected to cost roughly $300 billion over the next 20 years [7], the industry has an unprecedented opportunity to re-envision wastewater streams as resource-rich sanitation media. In particular, treatment strategies enabling nutrient recovery as well as energy recovery and generation should be advanced, enabling resource positive sanitation

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- the management of wastewater as a renewable resource for nutrient recovery and net energy production [16, 18, 58] that can have a net benefit for the environment [59] - to gain traction at a broad scale.

In response to these challenges, a great deal of research has been conducted on alternative wastewater treatment technologies that recover or produce energy during wastewater treatment. Most notably, anaerobic technologies can recover usable energy from organic carbon (typically measured as chemical oxygen demand, COD), and phototrophic technologies can increase the chemical energy of a wastewater through $CO_2$ fixation during growth and carbon storage. In addition to the production of bioenergy products such as methane, hydrogen, or electricity [29, 60–62], anaerobic processes are expected to be less energy intensive than aerobic processes due to a lack of aeration and a reduction in sludge wastage [26]. Although published studies have analyzed the performance of one or a small number of anaerobic system designs [29, 31, 63–66], an in-depth comparison of technologies focusing on their potential in domestic wastewater management is still needed. The limited literature on anaerobic treatment of domestic wastewater is partially due to lower methane production, lower COD removal, and higher methane solubility, all of which stem from the fact that domestic wastewater is a relatively dilute resource stream [29]. Ultimately, it is unclear whether the conversion of organic carbon to usable energy will be adequate for energy positive treatment using existing and emerging technologies [55, 67].

Beyond COD, nutrient (nitrogen, N; phosphorus, P) limits are becoming increasingly common across the U.S. [68, 69]. In addition to high capital costs of wastewater treatment plant (WWTP) upgrades (e.g., $3.36-3.96 billion for the for plants discharging to the Chesapeake Bay watershed [8]), the addition of biological nutrient removal (BNR) incurs higher operational costs that create or exacerbate financial challenges for utilities [70]. As an alternative to conventional BNR processes that leverage chemotrophic bacteria, phototrophic processes rely on light (typically sunlight) to promote growth and nutrient assimilation. As the phototrophs (including algae and cyanobacteria) take up inorganic carbon and grow, they also take up N and P from the wastewater and achieve nutrient recovery via assimilation. Studies have examined the potential for energy production using algae (e.g., [71, 72]) and have even examined the potential for energy positive wastewater treatment [67],but such studies have been limited to single cultivation technologies. To our knowledge there has not been a comprehensive comparative assessment of existing and emerging phototrophic technologies as tools to enable energy positive domestic wastewater management. In fact, studies focusing on bioenergy feedstock production with phototrophs have been largely disjointed from the wastewater literature, often using high strength synthetic media for cultivation (e.g., [73, 74]). The prospect of using phototrophs for nutrient recovery and/or bioenergy feedstock production holds significant promise, however, and warrants further discussion.

As researchers attempt to balance the potential gains in net energy production with performance and economic tradeoffs of each technology, the range of configurations for anaerobic and phototrophic systems continues to grow. To better understand the status and relative potential of each configuration, we undertook a critical literature review to characterize the demonstrated energy production by and critical barriers to a range of anaerobic and phototrophic technologies that have the potential to contribute to energy positive wastewater management. Based on available data, we quantified the typical performance of technologies in terms of treatment efficacy and bioenergy (or bioenergy feedstock) production, including the production of methane, hydrogen gas, electricity, biocrude oil, biodiesel, and heat. Seeking a deeper understanding of the potential energy balance of each technology, we also quantified usable energy yield (based on downstream conversion of bioenergy feedstocks) and anticipated energy consumption (based on experimental conditions in published studies). To be considered energy positive, a wastewater treatment scheme was required to produce energy in excess of the energy required to operate the process while also discharging water that meets regulatory standards. Given these constraints, we identify technologies with the greatest potential to enable energy positive carbon, nitrogen, and phosphorus management and present operational and performance targets for anaerobic and phototrophic treatment technologies to improve their net energy balance.

## 2.2 Anaerobic Systems

Anaerobic processes for bioenergy production are most commonly leveraged for industrial wastewater treatment or for solids management at domestic wastewater treatment facilities [26]. Limited application of these systems stems from a perceived need for high organic loading rates and mesophilic temperatures [61]. However, anaerobic technologies have been demonstrated at psychrophilic temperatures [33,61,75] and have the potential to be applied more broadly for low-strength wastewater treatment [58]. Anaerobic wastewater treatment processes can generally be categorized as suspended growth, sludge blanket, attached growth, membrane-based, or microbial electrochemical systems [26,76]. The first step of COD degradation in anaerobic treatment systems is the fermentation of complex organic matter into long chain volatile fatty acids, carbon dioxide, and hydrogen by acidogenic microorganisms. Long chain fatty acids are then further fragmented into acetic acid and hydrogen. Methane ($CH_4$) and hydrogen gas ($H_2$) are possible bioenergy products from anaerobic systems. In methane-producing reactors, acetoclastic methanogens ferment acetic acid to methane and carbon dioxide and hydrogenotrophic methanogens convert hydrogen and carbon dioxide to methane [77]. In microbial electrochemical technologies, exoelectrogenic bacteria oxidize acetate to carbon dioxide and produce electrical current transferring electrons to a conductive surface [78]. The anaerobic systems considered in this review are described below.

### 2.2.1 Suspended Growth Processes

Suspended growth processes are characterized by complete-mix conditions to prevent biomass from settling and to facilitate contact between the microorganisms and the wastewater. The most common processes include anaerobic sequencing batch reactors (ASBRs), completely-mixed anaerobic digesters, and the anaerobic contact process [26]. Of these, only ASBR had adequate peer-reviewed data (i.e., >5 papers) on the treatment of domestic wastewater.

#### Anaerobic Sequencing Batch Reactor (ASBR)

The ASBR progresses through four stages similar to the aerobic sequencing batch reactor: settle, decant, feed, and react [79]. ASBRs often have higher solids residence times (SRTs) compared to continuous flow processes and enable more precise operational parameter (e.g., hydraulic retention time, HRT) control [80]. However, their suitability for the treatment of low-strength wastewaters has been questioned due to low gas production on dilute streams, although intermittent mixing has been suggested to improve gas-liquid separation and to enhance sludge settling [81].

### 2.2.2 Sludge Blanket Processes

Successful operation of anaerobic sludge blanket processes relies on the aggregation of organisms into diversely populated granules capable of settling [82]. The granules form naturally from reactor operation and consist of a mixed population of bacteria and archaea that are able to carry out the overall fermentation and gas production from organic carbon substrates [25]. The gas bubbles produced from methanogenesis help to fluidize the granules, enhancing mass transfer without mixing [25]. Technologies with adequate (>5) peer-reviewed studies included the upflow anaerobic sludge blanket (UASB) and the anaerobic baffled reactor (ABR).

#### Upflow Anaerobic Sludge Blanket (UASB)

In a UASB reactor, wastewater enters the reactor and is distributed across the bottom, traveling upward through the sludge blanket [21]. Granular sludge in the reactor allows for high volumetric COD loadings (as compared to other wastewater treatment technologies) [26]. To enable better solids capture and to prevent loss of granules, modifications to the basic design have added packing material or a settling tank [26]. UASBs are advantageous because of their simple construction, scalability, and small footprint, though downstream processing is usually necessary to reduce effluent particulate organics and nutrient concentrations [65].

#### Anaerobic Baffled Reactor (ABR)

The ABR utilizes a sequence of baffles to impede the flow of wastewater as it passes through the reactor [83]. Flow patterns and gas production force sludge in the reactor to rise and settle slowly [26]. Since its inception, the ABR has undergone several modifications in an effort to improve performance, such as changes to baffle design, including a settler in the system, or achieving solids capture using packing material. Advantages of this process include: simplicity of construction and operation, prolonged retention of influent solids, staged operation, and insensitivity to shock loads [26]. Disadvantages include having to construct shallow reactors to accommodate gas and liquid upflow velocities as well as difficulty with distributing the influent flow evenly [63].

### 2.2.3 Attached Growth Processes

Attached growth anaerobic technologies rely on packing material in the reactor to provide surfaces for biofilm formation. The primary characteristics that differentiate reactors within this category are the packing material type and degree of bed expansion [26]. For example, the packed bed and fluidized bed configurations are operated at increasing upflow velocities, with fluidized bed being the highest. Because of the similarity of packed beds to the UASB and the availability of data for the fluidized system, only the anaerobic fluidized bed (AFB) was included.

#### Anaerobic Fluidized Bed (AFB)

AFB reactors are operated at high upflow velocities in order to suspend particulate media such as granular activated carbon (GAC) in the reactor [55], with wastewater treatment achieved by biofilms attached to the media. While AFBs are particularly effective for low strength wastewaters, the main shortcoming is minimal solids capture [26]. AFBs are therefore more appropriate for wastewater streams with primarily soluble COD.

### 2.2.4 Membrane-Based Processes

Membranes have been used in water treatment for over half a century, and are becoming increasingly common in applications ranging from wastewater treatment to desalination [84]. Microfiltration and ultrafiltration membranes are primarily used for particulate removal and can be arranged as flat sheets or hollow fibers [84]. One of the main benefits of using membranes in biological treatment processes is the completely independent control of SRT and HRT; SRT values have been reported as high as 300 days, where biomass was only removed from the system during sampling [33].

Anaerobic Membrane Bioreactor (AnMBR)

An AnMBR is an anaerobic reactor coupled with membrane filtration [29]. The membrane can be configured as external cross-flow, internal submerged, or external submerged [31]. Inclusion of a membrane allows for robust solids capture while also improving effluent quality over other mainstream anaerobic processes [26]. This increase in quality comes about because of the decoupling of SRT from HRT. Higher SRTs correlate to greater volatile fatty acid (VFA) and soluble COD removals [26]. Additionally, AnMBRs allow for a much smaller footprint by enabling higher solids concentrations in the reactor.

### 2.2.5 Microbial Electrochemical Technologies (METs)

METs (also referred to as bioelectrochemical systems, BES) leverage microorganisms capable of extracellular electron transfer [62, 85] to produce electrical energy from wastewater. Like all electrochemical technologies, such as fuel cells and batteries, METs are composed of an anode, where electrons for current are generated, and a cathode, where electrons are consumed. In METs, anaerobic bacteria naturally present in most wastewaters oxidize biodegradable organic matter and continuously transfer electrons to the anode [86]. Electrons flow from the anode, through an external circuit, to a cathode, where electrical current is consumed in a reduction reaction [62]. Current production in METs is dependent on the redox potential difference between organic matter oxidation at the anode ($E_o$ = -0.32 V) and current consumption at the cathode [87]. If the anode is more negative than the cathode, as in the case with oxygen reduction ($E_o$ = 0.82 V) in a microbial fuel cell (MFC), electrical current production is spontaneous. If the cathode reaction occurs at a redox potential that is more negative than the anode, such as hydrogen production ($E_o$ = -0.414 V) in a microbial electrolysis cell (MEC), then additional cell voltage must be applied to drive current production in the cell [62, 88]. Although a multitude of cathodic reactions (e.g., caustic production and hydrocarbon electro-synthesis [76]) have been paired with anodic oxidation of organic matter, the review will only focus on electricity and hydrogen production in METs.

### Microbial Fuel Cell (MFC)

The most commonly investigated MFC architecture is a single chamber reactor in which both the bio-anode and oxygen reduction cathode operate in the same solution [86]. The cathode electrode, which acts as the barrier between reactor solution and air, is coated with hydrophobic diffusion layers to allow oxygen transport but prevent water loss [89]. Although a variety of wastewaters have been evaluated for electricity generation [90], power production has been significantly lower ($< 0.5$ W$\cdot$ m$^{-2}$ cathode area) than reactors fed synthetic and well buffered solutions (1.0-4.3 W$\cdot$ m$^{-2}$) due to low solution conductivity as well the dilute concentrations and complex nature of organic substrates in domestic wastewater [76, 91–93]. Additionally, cathodic materials are often expensive due to the need of precious metals (e.g., platinum) [94, 95].

### Microbial Electrolysis Cell (MEC)

MECs produce hydrogen from substrate by coupling a hydrogen evolution electrode to the bio-anode [60, 96–98]. Hydrogen is a promising fuel for meeting future energy demand because it only produces water when combusted or oxidized in a fuel cell and has a high energy yield (142.35 kJ$\cdot$ g$^{-1}$) [99]. Since MEC current production is not spontaneous, voltage must be applied to produce hydrogen (0.6-1.2 V in practice [88]). Due to cathode catalyst, electrolyte, and substrate deficiencies, energy consumed by applying voltage can exceed the energy recovered as hydrogen gas [99, 100]. Also, to prevent hydrogen losses due to hydrogenotrophic methanogenesis that occur in single chamber architecture [101, 102], a membrane or gas diffusion electrode is required to separate anode and cathode [97, 100].

## 2.3 Phototrophic Systems

Simple, passive phototrophic processes (cultivating algae and/or phototrophic bacteria) such as open ponds are commonly used to treat municipal and agricultural wastewaters [103]. To date, the objective for these technologies tends to be nutrient (and often COD) removal from wastewater, rather than nutrient recovery or bioenergy feedstock production. Alternatively, more capital-intensive systems such as photobioreactors have been studied for phototroph cultivation, but this work has most often focused on bioenergy feedstock cultivation rather than wastewater treatment (e.g., [104, 105]). Both types of systems predominantly operate with suspended cultures in open (e.g., ponds [106]) or closed systems (e.g., photobioreactors [20]) that allow for sunlight penetration and nutrient assimilation to promote energy and growth before biomass is harvested [107]. Alternative systems consist of attached or immobilized phototrophs for easier harvesting [108]. Ultimately, the energetic benefit of phototrophic systems stems from the fact that they can increase the energetic content of wastewater through the conversion of light energy to chemical energy (as organic carbon). In order to evaluate the relative potential of phototrophic technologies in achieving energy positive municipal wastewater treatment, only published studies using actual wastewater as the growth medium have been included in the analysis.

### 2.3.1 Suspended Systems

Conventional phototrophic systems consist of suspended cultures that are operated in either continuous, batch, or semi-batch mode [109, 110]. The most common large-scale phototroph cultivation systems are waste stabilization ponds (WSPs) [103], high rate algal ponds (HRAPs) [106], stirred tank reactors [111], and tubular photobioreactors (PBRs) [20, 112] . At laboratory-scale, a wider variety of reactor configurations have been evaluated, including flat panel (a.k.a., flat plate) and annular PBRs [104], as well as more basic well-mixed systems that are simply lit from overhead (these studies were classified as "Stirred Tank Reactors" for this review) [113].

#### High Rate Algal Pond (HRAP)

While open raceway ponds are used commercially for the production of algal biofuels and health products [114], a subset of published studies use HRAPs for wastewater treatment (e.g., [106, 115]). HRAPs are open raceway ponds first proposed in the 1950s with the goals of providing improved wastewater treatment over traditional WSPs and algal biomass for potential biofuel applications [116]. Although they have the potential to be a more cost effective solution than PBRs for wastewater treatment [117, 118], HRAPs have relatively low biomass productivity (and thus require larger land areas) as compared to reactor-based technologies [20].

Photobioreactor (PBR)

Another widely used technology for cultivating algal biomass is the PBR [104, 119]. These closed array systems allow for high biomass productivity as well as axenic growth conditions for monoculture maintenance [120]. Although a range of configurations have been evaluated at the lab-scale [104, 119], larger systems tend to be tubular PBRs due to economies of scale. There are relatively few studies that examine PBRs in conjunction with wastewater treatment, largely because of high costs compared to other treatment technologies [119] and because axenic cultures are generally not targeted for municipal wastewater treatment. Most PBR studies focus on pure species with high lipid productivities and, consequently, higher energy potential and revenue generation [120–122]. Given the objective of this study, only those published studies using PBRs fed wastewater media were included.

Stirred Tank Reactor

There is extensive literature on phototrophic growth in stirred tank reactors (open, completely mixed reactors lit from overhead). Although published studies using stirred tank reactors cover a range of operational conditions (including various lighting schemes, batch vs. continuous vs. semi-continuous operation, etc.) and a subset have been performed at the pilot-scale [67] the majority of these studies have been at the laboratory-scale (e.g., [123–125]). In order to look for general trends in performance of stirred tank reactors, data from these studies have been aggregated to identify performance trends and enable comparisons to larger-scale, more broadly applied technologies (e.g., HRAPs). Any insights gained may be applied to the design of larger-scale batch or sequencing batch reactors for both wastewater treatment and algal biomass production.

Waste Stabilization Pond (WSP)

WSPs are the most widely used phototrophic treatment technology [126]. In the U.S. alone there are >7,000 WSPs in use, which accounts for over one-third of all centralized treatment systems [103]. During the day, phototrophs in these systems produce dissolved oxygen, which facilitates COD degradation by aerobic heterotrophs [127] and promotes photo-oxidative damage for pathogen removal [128–130]. Although WSPs are often a cost effective solution for wastewater management utilities [131], they are used almost exclusively in rural areas due to large land requirements [132]. With the exception of early visionary proposals linking wastewater to bioenergy with algae [116, 133–135], WSP literature has focused almost exclusively on wastewater treatment (removal of COD, N, P, heavy metals [136, 137]) with little discussion of biomass production or potential biofuel applications. Despite limited literature linking WSPs to bioenergy feedstock cultivation, this technology represents one of the easiest opportunities to transition from an existing energy neutral/consuming technology to an energy producing process given that algal biomass is already generated.

### 2.3.2 Attached Growth Systems

The cost of biomass harvesting (including flocculation, centrifugation, and sedimentation [107]) remains a key barrier to the broad implementation of suspended growth algal systems [113]. Although sedimentation is often the most inexpensive approach, it achieves low (50-90% [106, 138]) biomass recoveries and is typically used when low value biomass is being removed from the system [139]. Technologies that seek to achieve high percentages (>95%) of suspended biomass recovery for use as biofuel feedstock would add significantly to the cost of operation [106, 140, 141]. As an alternative to suspended growth, attached growth systems restrict algal growth to physical structures resulting in aggregated biomass that either sloughs off the structures or can be removed through cleaning [142]. While there are a number of different attached growth systems available (e.g., Algaewheel and other industrial solutions [143] as well as various immobilized gel matrices [144]), the data necessary to perform the energetic analysis for most attached growth technologies was lacking. One exception was the algal turf scrubber (ATS), which has been the focus of a number of studies and which reported adequate data for its inclusion in this study [145, 146].

Algal Turf Scrubber (ATS)

ATSs consist of long, inclined beds typically constructed of landfill liner that support mixed community biofilms that include cyanobacteria, filamentous periphyton, and epiphytic diatoms [142, 147, 148]. As water flows down the beds into a concrete sump, nutrients are taken up by the biofilm, supporting microbial activity and reducing the concentrations of nutrients in the effluent [149]. When biomass accumulates, harvesting is often performed by machinery (such as a loader) driven across the bed [142]. Although it is not a common process, there are several private companies operating ATSs on a large scale, notably, Aquafiber Technologies (7.5 million gallons per day [MGD]) and HydroMentia (capacity 30 MGD) [143].

## 2.4 Data Analysis

### 2.4.1 Criteria for Inclusion in This Study

A comprehensive literature review was conducted on the technologies listed above, with a focus on studies demonstrating treatment of municipal-strength wastewaters (COD <500 g-COD·m$^{-3}$). For phototrophic systems, comparison studies have found that synthetic wastewater, though displaying comparable nutrient removal rates, generates more biomass than wastewater-based studies [110] and was thus excluded from this review. For methane producing systems, studies using synthetic wastewater with relevant COD concentrations were included because differences in performance (between synthetic and real wastewater) were not readily observed. For METs, all studies with influent COD <500 g-COD·m$^{-3}$ used real wastewaters (i.e., all studies that used synthetic wastewater had COD values above 500 g-COD·m$^{-3}$ and were thus excluded). Once relevant studies were identified, many were excluded from further analysis due to insufficient data that prevented the calculation of energy production normalized to contaminant removal. If a required value was not explicitly stated but prerequisite values were given, the unknown values were calculated (see Figures A.1-A.2 for inclusion/exclusion decision-making). Ultimately, these data were used to report the effluent COD and energy (as kilojoules, kJ) recovered by anaerobic treatment as well as effluent N/P and energy produced by phototrophic technologies (Figure 2.1).

### 2.4.2 Energetic Analysis

Anaerobic Technologies

Anaerobic technologies recovered energy either in the form of methane (mol-$CH_4$·g-$COD^{-1}$), hydrogen gas (mol-$H_2$·g-$COD^{-1}$), or electricity (kJ·g-$COD^{-1}$). In order to compare the data objectively, each was normalized to kJ recovered per g-COD removed by converting each energy source to kJ using standard conversion factors based on energetic content: 803 kJ·mol-$CH_4^{-1}$ [26], 286 kJ·mol-$H_2^{-1}$ [150], and by converting electricity (reported in kWh) to kJ by multiplying by 3,600 sec·$hr^{-1}$ (Equations A.1-A.3). Results for each technology were compared on the basis of per capita and per $m^3$ of wastewater treated using the conversions discussed in Section 2.4.3

Phototrophic Technologies

Phototrophic technology data were compiled from articles that reported both biomass generated and nutrient (N and/or P) removal. Biomass was either reported as total, maximum, or average VSS (g·$m^{-3}$), as productivity (g·$m^{-3}$·$day^{-1}$), or as aerial productivity (g·$m^{-2}$·$day^{-1}$). The SRT, experiment length, and reactor volume were leveraged to convert all numbers to an average daily productivity (g·$m^{-3}$·$day^{-1}$). Biomass productivities were then normalized by the average nutrient removal per day (g·$m^{-3}$·$day^{-1}$) to achieve g-biomass produced per g-nutrient removed from the treatment system. To convert biomass productivity to energetic potential, reported VSS were converted to units of COD (see Table A.2). Two scenarios were considered using macromolecule content (lipids/-carbohydrates/proteins, L/C/P) within typical ranges from the literature [151–153]: a low COD/VSS ratio of 1.47 g-COD·g-$VSS^{-1}$ (assuming 10/40/50% lipids/carbohydrates/proteins, L/C/P [111, 125, 151–153]) and a high COD/VSS ratio of 1.84 g-COD·g-VSS$^{-1}$ (assuming a ratio of 30/20/50% L/C/P [111, 151–153]). COD calculations were performed assuming lipids could be represented as stearic acid ($C_{18}H_{36}O_2$), carbohydrates as glucose ($C_6H_{12}O_6$), and proteins as $C_{16}H_{24}O_5N_4$ [154]. Although higher COD/VSS ratios would be possible if higher lipid content were achieved (e.g., 70% lipids [20]), the ratios used here represent a reasonable range of expected compositions [111, 125] to avoid overly optimistic ratios that would artificially increase calculated energy yield. Although it has been reported that some species can obtain greater than 80% lipids by dry biomass weight [155,156], mixed algal wastewater cultures routinely see far less lipid accumulation, with an average around 10% [111, 125]. Once biomass productivities were converted to COD, the energetic potential of the biomass was then calculated using a theoretical value of 13.9 kJ·g-COD$^{-1}$ [157]. Results for each technology were compared on a per capita basis as described in Section 2.4.3.

Conversion to Usable Energy

Although the energetic content of treatment system products may provide insight into the fundamental limitations of a given technology, the question regarding the feasibility of energy positive treatment can only be answered by determining the usable energy (e.g., electricity, heat, liquid fuel) provided by each treatment system. For anaerobic systems, the outputs include methane, hydrogen, and electricity. Given that the predominant form of energy consumed by treatment plants is electricity, methane and hydrogen were converted to electricity in a fuel cell at a 42.3% conversion efficiency [158].

In order to predict the production of usable energy from phototrophic biomass, the energy yield from four different conversion processes - hydrothermal liquefaction (HTL), transesterification, anaerobic digestion, and combustion - were also calculated. Although anaerobic digestion has a long history in the conversion of algal biomass to methane [134, 135, 159], direct combustion of algal biomass has been proposed as more energetically favorable than converting biomass to any biofuel [67, 160]. For the conversion of phototrophic biomass into liquid fuels, both transesterification and HTL were considered, with HTL representing an emerging process of interest to the algae-to-biodiesel community [161, 162]. HTL has been applied to wastewater-grown biomass (e.g., [163, 164]), although energy balances have identified biomass harvesting and dewatering as key barriers to achieving energy positive systems [67]. The list of assumptions and values used for these calculations can be found in Table A.2.

Energy Consumption

An estimation of energy consumption for each technology was included in order to evaluate the feasibility of net energy positive wastewater treatment. However, the published studies analyzed did not include energy consumption data with the exception of Sturm and Lamer 2011. In order to quantify energy consumption of each process, the energy demand from various activities (e.g., pumping, mechanical mixing, gas sparging, etc.) [26, 165] was estimated using standard design equations and the published range of design and operational parameters (see Section A.4 for a detailed explanation).

### 2.4.3 Unit Conversions and Efficiency Calculations

Data were normalized and reported in one of four ways: as energy per gram of pollutant removed, energy per capita, energy per cubic meter of wastewater treated, and as a percent of energetic potential recovered. Energetic data normalized to pollutant removal (kJ per g-COD, g-N, or g-P) was calculated directly from the published data sets included in the review. These data (in units of $\text{kJ}\cdot\text{g-pollutant removed}^{-1}$) were then normalized to per capita values by multiplying (i) by the average percent removal of that pollutant by a given technology, and (ii) by the average daily per capita production of that pollutant (180 g-COD; 13 g-N; 2.1 g-P [166]). Next, energy productions were also reported per cubic meter of wastewater treated by assuming a wastewater production rate of $0.36\ \text{m3}\cdot\text{person}^{-1}\cdot\text{day}^{-1}$ resulting in a wastewater composition of $500\ \text{g-COD}\cdot\text{m}^{-3}$, $36\ \text{g-N}\cdot\text{m}^{-3}$, $5.8\ \text{g-P}\cdot\text{m}^{-3}$. For efficiency calculations (e.g., percent of chemical energy recovered), COD was assumed to contain roughly $13.9\ \text{kJ}\cdot\text{g-COD}^{-1}$ [157], resulting in an influent energetic content of 7,000 $\text{kJ}\cdot\text{m}^{-3}$. This conversion factor is lower than more recent values reported in the literature

(17.7-28.7 kJ·g-COD$^{-1}$ [167]), but was used throughout the manuscript to provide a consistent framework for energy conversions. All energy values (in units of kJ) represent the energetic content of produced fuel (methane, hydrogen, or electricity) for anaerobic systems or produced biomass for phototrophic systems, unless otherwise noted (fuels are converted to electricity; biomass is converted to heat, methane, biodiesel, and biocrude oil).

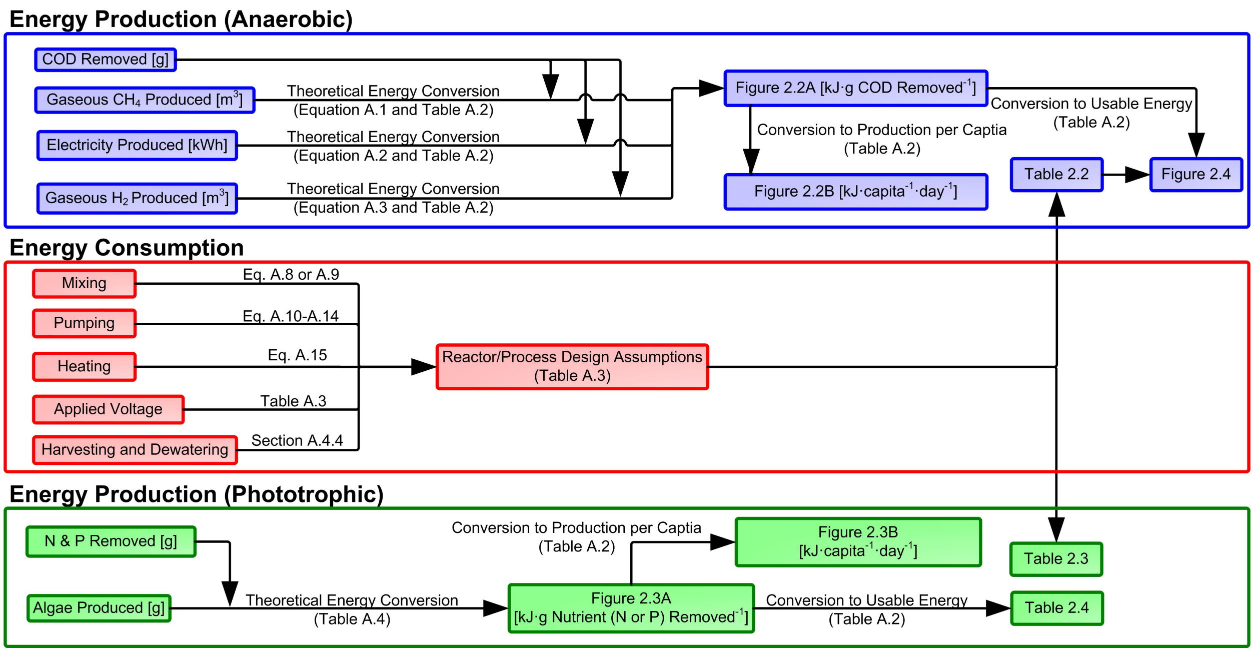


Figure 2.1: Flow chart showing the process of data acquisition and analysis used in the manuscript along with conversion factors and their location in Appendix A.

## 2.5 Results and Discussion

In the review of the peer-reviewed literature a total of 225 anaerobic and 86 phototrophic papers were screened and assessed according to the inclusion criteria (Figures A.1,A.2). Of the papers reporting on anaerobic technologies, only 32 met the necessary criteria for energetic and treatment analysis with a total of 122 experimental data sets (i.e., if the study reported multiple experimental conditions or replicates, all that met the inclusion criteria were included in this review). Published data on phototrophic technologies were less complete, with only 23 papers meeting the necessary criteria for treatment analysis with a total of 33 and 58 data sets for N and P removal, respectively. Of these papers, 13 had the necessary biomass productivity for energy analysis, resulting in 21 and 25 experimental data sets for energy production per g-nutrient (N or P) removed (across 37 independent data sets). Furthermore, 9 of these 37 datasets were excluded because they reported greater than 50 g or 225 g of algal biomass grown per g-N or g-P removed, respectively, which was deemed to be outside the likely range of feasible biochemical compositions. Finally, WSPs were excluded from the energetic analysis due to a lack of biomass productivity data.

### 2.5.1 Energetic Analysis

The energetic analysis began by determining fuel (anaerobic) or bioenergy feedstock (phototrophic) production from each study and the associated caloric content (Section 2.5.1). Energy consumption (Section 2.5.1) of each technology was then estimated based on experimental conditions in published studies and on additional assumptions detailed in Section A.4. An energy balance between consumption and production was then detailed for anaerobic systems to estimate net energy given typical experimental conditions in order to identify key barriers to energy positive treatment (Section 2.5.2). An energy balance was excluded for phototrophic technologies because of the uncertainty associated with downstream conversion to usable fuels, but available data were leveraged to set targets for cultivation and downstream fuel conversion processes (discussed in Section 2.5.3). Lastly, we examined the dichotomy between emerging (energy production) and traditional (effluent quality) objectives for treatment technologies (Section 2.6).

#### Energy Yield

The average energy recovery by anaerobic systems ranged from 0.48 kJ·g-COD$^{-1}$ (MFC) to 7.3 kJ·g-COD$^{-1}$ (ABR) and was highest for gas producing technologies (Figure 2.2a). The average percent energy recovery (as methane, hydrogen, or electricity) from degraded COD by each technology was as follows (from greatest to least; average  standard deviation): ABR (47.5 ± 4.5%), AnMBR (35.4 ± 26.8%), AFB (33.8 ± 12.9%), UASB (24.0 ± 11.4%), ASBR (17.7 ± 10.1%), MEC (14.3 ± 14.4%) and MFC (1.6 ± 1.4%). When including typical percent COD removals for each technology, this range would equate to roughly 40-

1,200 kJ·capita$^{-1}$·day$^{-1}$ (Figure 2.2b) or 110-3,300 kJ·m$^{-3}$ of treated wastewater. After conversion of gases to usable electricity in a fuel cell (42.3% efficient), these values represent recoveries of roughly 2-20%. UASBs and AnMBRs had the two highest reported energy recovery data sets (12.2 and 9.7 kJ·g-COD$^{-1}$ degraded, respectively), but AnMBRs also had the greatest variability (standard deviation of 4.3 kJ·g-COD$^{-1}$). The energy recovery by MECs was statistically different from most of the methane-producing technologies (p-values $<0.024$, $\alpha=0.05$; two-tailed, unpaired t-test) except ASBRs (p-value = 0.077), which could not be shown to be statistically different. MFCs did, however, exhibit significantly lower energy production (p-value = 0.048) with average per capita energy recovery 5-15 fold lower than gas producing technologies, or 2.3-6.5 fold lower after gas conversion to electricity. Although MFC power production from wastewater was limited by substrate conductivity and strength, power densities from single chamber MFCs fed optimized synthetic solutions ($\sim$1.4 kJ·g-COD$^{-1}$) [92] would have still been only 19-55% of the average reported energy recovery rates for methane-producing technologies.

Although discussions linking energy and nutrients in wastewater are generally focused on potential fertilizer offsets from nutrient recovery (e.g., [18]), the average energetic content of cultivated phototrophic biomass across all technologies (kJ·capita$^{-1}$·day$^{-1}$) was 130-510% of the energy saved from offsetting fertilizer production (assuming 100% nutrient recovery and 69 kJ·g-N$^{-1}$ [168] and 7.6 kJ·g-P$^{-1}$ [168] for synthetic fertilizer production). The average bioenergy feedstock production by phototrophic systems ranged from 210 kJ·g-N$^{-1}$ (HRAP) to 760 kJ·g-N$^{-1}$ (PBR), and 640 kJ·g-P$^{-1}$ (PBR) to 2,500 kJ·g-P$^{-1}$ (stirred tank) (see Figure 2.3a and Section A.3.2 for energy production values for each technology). On a per capita basis, the average energy production for each technology was as follows (if both N and P data were available in a given data set, the lesser prediction of biomass production based on per capita N and P was used): stirred tank reactor (4,700 $\pm$ 3,200 kJ·capita$^{-1}$·day$^{-1}$), ATS (2,300 $\pm$ 1,100 kJ·capita$^{-1}$·day$^{-1}$), HRAP (1,800 $\pm$ 860 kJ·capita$^{-1}$·day$^{-1}$), and PBR (1,200 $\pm$ 340 kJ·capita$^{-1}$·day$^{-1}$) (Figure 2.3b). This range would equate to 3,400-13,000 kJ·m$^{-3}$ of treated wastewater, or 280-400% of the potential recovery from methane-producing anaerobic technologies. As expected, the ratio of energy yield to P uptake was 2-14x higher as compared to N uptake (except for PBR data). One explanation for the low energy potential per gram of P in PBRs is the low N:P removal ratio reported by the majority PBR studies (all of which had influent N:P ratios less than 1). For HRAP, stirred tank reactors, and ATS systems, the ratio of N to P mass uptake was roughly 7.5 $\pm$ 3.0 (average $\pm$ standard deviation). These values are higher than typical assumptions of biochemical composition of microalgae using an N:P mass ratio of 4.5:1 (N:P molar ratio of 10:1; e.g., [169]), but within the range that algae can adapt their N:P ratio (reported mass ratios range from 2.3-45:1 [170]). In the case of PBR experiments, data analysis was limited to two wastewater-relevant studies with adequate data. Additionally, it is possible that low ratios of biomass production per P removed were partially the result of alternate mechanisms (other than growth) including luxury uptake of P (microalgae have been shown to accumulate up to ~3x normal cellular P [171]) and P adsorption to cell surfaces [172]. Additional sources of feedstock production variability may have included carbon limitation, reactor and process design, and/or differences in lighting efficiencies. Nutrient rich phototrophic systems are often carbon limited owing to a C:N molar ratio typically less than cell requirements (~3:1 in typical wastewater vs. 6:1 cellular) [26, 173, 174].

Table 2.1: Range and average percent COD or nutrient (N or P) removal for each technology used in Figures 2.2 and 2.3.

| **Technology** | **Average Percent Removal (min,max)** | |
|---|---|---|
| | COD | |
| ASBR | 58.1 (33, 91) | |
| UASB | 67.6 (54, 85) | |
| ABR | 90.3 (88.7, 92.5) | |
| AFB | 82 (72, 89.7) | |
| AnMBR | 86.7 (82, 90) | |
| MEC | 78 (33.7, 96.7) | |
| MFC | 45.5 (19, 83) | |
| | N | P |
| HRAP | 67.1 (36, 87.2) | 52.1 (32, 72.9) |
| PBR | 78.5 (68, 89.7) | 93.2 (85, 99) |
| Stirred Tank | 62.3 | 78.2 (7, 100) |
| ATS | 70.5 (18.1, 90.7) | 78.6 (58.3, 95.7) |

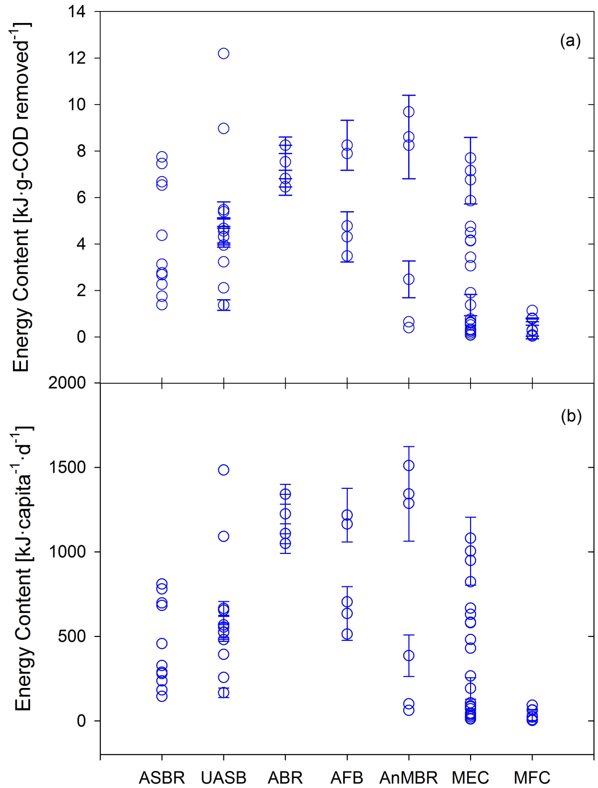


Figure 2.2: (a) Energy content [kJ-fuel·g-COD removed$^{-1}$] for each paper studying anaerobic technologies with an influent COD below 500 g·m$^{-3}$ (for synthetic wastewater) or using actual domestic wastewater. (b) Energy content [kJ-fuel·capita$^{-1}$·d$^{-1}$] determined by multiplying values from Figure 2.2a by 180 g-COD·capita$^{-1}$·d$^{-1}$ and by the respective average percent COD removals for each technology (Table 2.1). All energy products (methane, hydrogen, electricity) are reported as kJ using theoretical unit conversions (see Appendix A). Individual points represent distinct experimental data sets, with error bars extending to ± standard deviation (if reported).

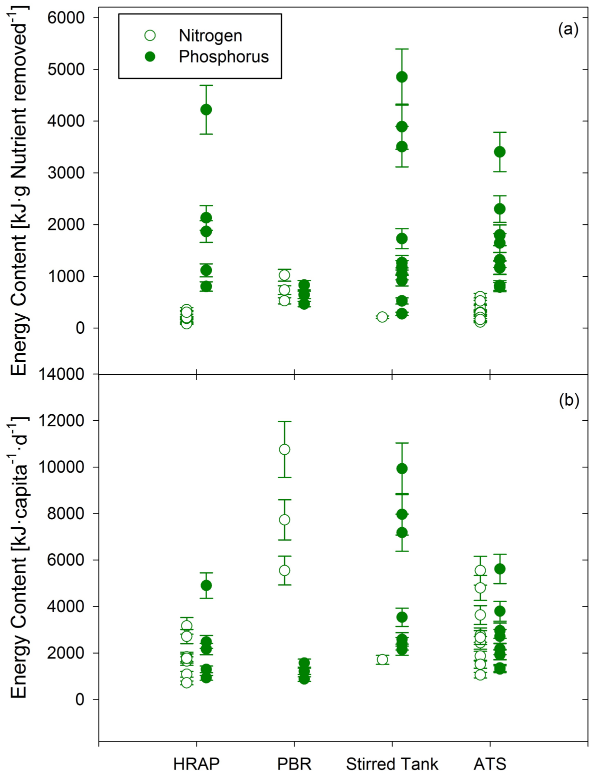


Figure 2.3: (a) Energy potential of phototrophic technologies [kJ-algal biomass·g-nutrient removed$^{-1}$] showing relative bioenergy feedstock production based on nutrient removed (N or P). (b) Energy content [kJ-algal biomass·capita$^{-1}$·d$^{-1}$] determined by multiplying values from Figure 2.3a by 13 g-N·capita$^{-1}$·d$^{-1}$ or 2.1 g-P·capita$^{-1}$·d$^{-1}$ and by the respective average percent N and P removals for each technology (Table 2.1). Energy products are reported as kJ using theoretical unit conversions (see Appendix A). Individual points represent distinct experimental data sets, with error bars extending from high to low COD/VSS assumptions (discussed in Section 2.4.2).

Energy Consumption of Anaerobic and Phototrophic Technologies

While the goal of this manuscript was to examine the potential for full-scale anaerobic and phototrophic technologies to achieve energy positive treatment, pilot- and full-scale data in the peer-reviewed literature was severely limited requiring the inclusion of laboratory-scale data. Recognizing that a full-scale plant would be operated differently (and likely more efficiently) than its laboratory-scale counterpart, analysis of laboratory-scale data may still offer meaningful insights to the major barriers to full-scale implementation that each technology faces. Therefore, the energy consumption analysis that follows is not argued to be a perfectly accurate representation of full-scale energy consumption, but rather a starting point for a discussion of how design and operational decisions will influence the ability of anaerobic and phototrophic technologies to achieve energy positive wastewater management. Estimated energy consumption was highly variable across technologies (Tables 2.2 and 2.3), due to the wide range of experimental conditions and operational controls (e.g., fouling prevention and mitigation). All systems were assumed to be gravity fed, with energy consumption resulting from any additional operational requirements (assumptions detailed in Section A.4). Since UASB, ABR, and MFC generally operate as passive systems, the operational energy consumption for these anaerobic technologies was minimal (assuming the reactor was not heated). Studies included in this analysis operated between 10-35°C, but it should be noted the decision to heat would require roughly 4,200 kJ·m$^{-3}$ per 1 °C increase above influent wastewater temperature; the magnitude of this energy demand underscores the importance of operation at ambient temperatures and the importance of developing anaerobic technologies capable of psychrophilic operation.

Table 2.2: Ranges of energy consumption for anaerobic technologies based on experimental data from examined literature (kJ·m$^{-3}$ wastewater treated).

| Technology | Mixing | Pumping | Heating | Applied Voltage |
|---|---|---|---|---|
| ASBR | 4,800-9,400[a] | 28-31[b] | | - |
| UASB | - | - | | - |
| ABR | - | - | | - |
| AnMBR | 42,000-58,000[c] | 36-120[d] | | - |
| AFB | - | 55-130[e] | 4,200[f] | - |
| MEC | - | - | | 2,800-7,900 |
| MFC | - | - | | - |

[a] Mechanical mixing (Section A.4.1 and Table A.3)
[b] Effluent pumping (Section A.4.5 and Table A.3)
[c] Biogas sparging (Section A.4.2 and Table A.3)
[d] Permeate pumping (Section A.4.5 and Table A.3)
[e] Recirculation pumping (Section A.4.5 and Table A.3)
[f] Energy required for each increase in °C (not included in final energy balance) (Section A.4.6 and Table A.3)

Although mechanical mixing and applied voltage result in appreciable energy consumption

Table 2.3: Ranges of energy consumption for phototrophic technologies based on experimental data from examined literature (kJ·$m^{-3}$ wastewater treated).

| **Technology** | **Mixing** | **Pumping** | **Harvesting**[a] |
|---|---|---|---|
| HRAP | 3.2-9.6[b] | - | |
| PBR | 6,300-13,000[c] | 55-58[d] | |
| Stirred Tank | 770-3,100[e] | 28-31[f] | 34-170 |
| WSP | - | - | |
| ATS | - | - | -[g] |

[a] Low value is coagulation-flocculation with belt press filter for dewatering, high value is gravity settling with centrifugation (Section A.4.4 and Table A.3)
[b] Paddlewheel mixing (Section A.4.3 and Table A.3)
[c] Aeration (Section A.4.2 and Table A.3)
[d] Influent lift pump (Section A.4.5 and Table A.3)
[e] Mechanical mixing (Section A.4.1 and Table A.3)
[f] Effluent pumping (Section A.4.5 and Table A.3)
[g] Although minimal energy would be required for the physical harvesting of algae from ATS, it was not estimated due to lack of available data.

for ASBR and MEC, respectively, the largest source of energy consumption among anaerobic technologies was gas sparging to manage membrane fouling in AnMBR. Estimates of energy consumption from biogas sparging were based on published rates from 0.67 [34] to 0.93 [33] $L_{Gas} \cdot L_{Reactor}^{-1} \cdot min^{-1}$, which represent very high rates of gas addition to reactors. Scale-up and more targeted gas scouring techniques can certainly reduce the gas flow demand [175], but the use of alternative approaches to fouling mitigation and prevention may be even more energetically favorable. In particular, external cross-flow AnMBR configurations [31] or staged reactors with media for biofilm attachment [44] may offer distinct advantages over submerged reactors, so long as the operational conditions are scalable and they mitigate fouling with less energy-intensive methods than gas scouring.

For phototrophic technologies, PBRs, which also rely on gas sparging for mixing, had the highest energy consumption. Typical sparging rates in PBRs are often 0.1-0.3 $L_{Gas} \cdot L_{Reactor}^{-1} \cdot min^{-1}$ [104, 176], with actual rates in the field dependent upon biomass characteristics and tendency to aggregate. Ultimately, however, mixing requirements for algal systems are less than many chemotrophic systems due to decreased cell aggregation and a higher sensitivity of algal cells to shear forces [177]. Passive systems such as WSP and ATS consume almost no energy during operation, as is also the case with HRAPs (which require very few paddle-wheels per hectare). These systems require much larger land areas [111], however, resulting in a distinct tradeoff between aerial productivity and energy consumption during cultivation.

### 2.5.2 Energy Balance & Treatment Efficacy of Anaerobic Technologies

#### Usable Energy Balance for Anaerobic Processes

Given that treatment processes are generally powered by electricity, the caloric content of the gaseous products from anaerobic processes were converted to electricity and compared with consumption (Figure 2.4). MFCs were the only technology evaluated that can directly produce electricity. The estimated electricity recovery from methane and hydrogen was assumed to be 42.3% for conversion of methane or hydrogen using fuel cells [158]. A significant amount of energy (nearly 60%) is lost in the conversion of alternative fuels to electricity used directly by treatment plants, which further limits the potential for energy neutral operation. The red boxes in Figure 2.4 represent energy consumption normalized to g-COD removed, excluding energy from heating. These values were calculated by converting the data from Table 2.2 (energy consumption per $m^3$ treated) to a COD removal basis using average COD removal efficiencies (Table 2.1) and an assumed influent concentration of 500 g-COD·$m^{-3}$. The energy demand for heating (included in Table 2.2) was excluded from the energy balance because many of the studies operated at ambient temperatures, and no trend was observed between operating temperature and energy recovery or production (data not shown).

Although all anaerobic technologies were capable of recovering energy, only four appear to be immediately capable of net energy positive operation: UASB, ABR, AFB, and MFC. It was assumed that three of these technologies (UASB, ABR, and MFC) could be operated as passive systems with no significant operational energy. For the remaining four technologies, energy consumption demands were exacerbated by COD removal efficiencies below 100% (e.g., ASBRs - 58.1% COD removal efficiency - require 1.7 $m^3$ for every 500 g-COD degraded). With ASBR energy recovery ranging from 1.4-7.7 kJ·g-$COD^{-1}$, the energy balance was hindered by the energy intensity of mechanical mixing (17-33 kJ·g-COD$^{-1}$) and, to a lesser extent, effluent pumping (0.10-0.11 kJ·g-COD$^{-1}$). To achieve energy neutrality, the energy for mixing must be drastically reduced. While AnMBRs achieved some of the highest energy recovery values (up to 86.7% with 0.4-9.7 kJ·g-COD$^{-1}$), continuous biogas sparging for mixing as well as to prevent and mitigate membrane fouling led to significant energy consumption (100-145 kJ·g-COD$^{-1}$). The gas flow rate needed for sparging would have to decrease by more than an order of magnitude to about 0.03 $L_{Gas} \cdot L_{Reactor}^{-1} \cdot min^{-1}$ (with no increase in TMP) or alternative strategies for fouling management would have to be developed in order for energy neutrality to be achieved. Alternatively, the divide between energy recovery and consumption could be narrowed if methane recovery from the effluent were improved (on the order of 30-50% of produced methane may be lost to the effluent [28, 33]). MEC energy consumption is a function of applied voltage and current production, and could be improved by developing cost effective low over-potential hydrogen evolution catalysts. However, operating any catalyst in wastewater will likely limit the kinetics of proton reduction. MECs have the highest energy recovery potential (based on the thermodynamics of hydrogen, methane, and electricity production; see Figure 2.4 and Section A.3.1) and recovered more energy than MFCs, but the energy consumed by applying a voltage make MECs less energetically favorable than MFCs for low strength wastewater treatment.

Efficacy of Anaerobic Technologies for COD Removal

To replace energy intensive aerobic processes, anaerobic technologies must balance energy production with efficient COD removal. Although limited data was available in many cases, a review of the literature revealed that COD removal was highest (80-90%) in systems that included physical separation of biomass from the effluent (AnMBRs; Figure 2.5d) or leveraged attached growth (AFBs; Figure 2.5c). The variability of COD removal in METs was the highest (Figure 2.5e), which can be partially attributed to reactor operation. Continuous flow METs, for example, achieved lower COD removals than batch-fed reactors. Comparison with energy consumption data shows that tradeoffs between net energy balance and effluent quality do exist in some cases (e.g., AnMBR), but alternative configurations may

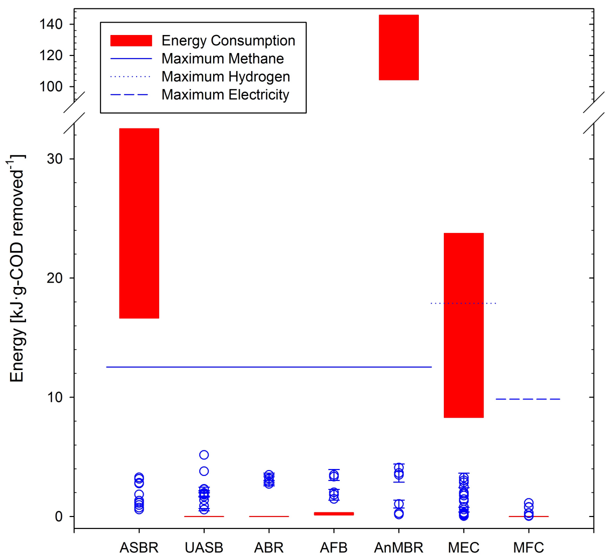


Figure 2.4: Energy recovery, consumption, and theoretical maximum energy yield for each technology. Blue circles represent energy production per gram of COD removed in experimental data sets from the literature. Red boxes - indicating the range of energy consumed that needs to be overcome for energy positive treatment (excluding heating requirements of the wastewater) - were calculated based on volumetric energy requirements (Table 2.2) coupled with typical COD removal of each technology (Table 2.1) and an assumed influent of 500 g-COD·m$^{-3}$. Blue horizontal lines show the maximum energy that can be generated for methane (solid), hydrogen (dotted), and electricity (dashed) based on thermodynamics (calculations shown in the Section A.3).

be able to achieve a high quality effluent under net energy positive operation (AFB). Since the carbon energy density of domestic wastewater is low, innovative solid-liquid separation methods will be needed to meet traditional treatment objectives and achieve energy positive COD removal.

### 2.5.3 Energy Balance & Treatment Efficacy of Phototrophic Technologies

#### Usable Energy Balance of Phototrophic Processes

In the conversion of phototrophic biomass to usable energy, HTL achieves the highest energy output followed by anaerobic digestion, combustion, and transesterification (maximum values in that order, Table 2.4; details of assumptions in Section A.4.8). Although biocrude

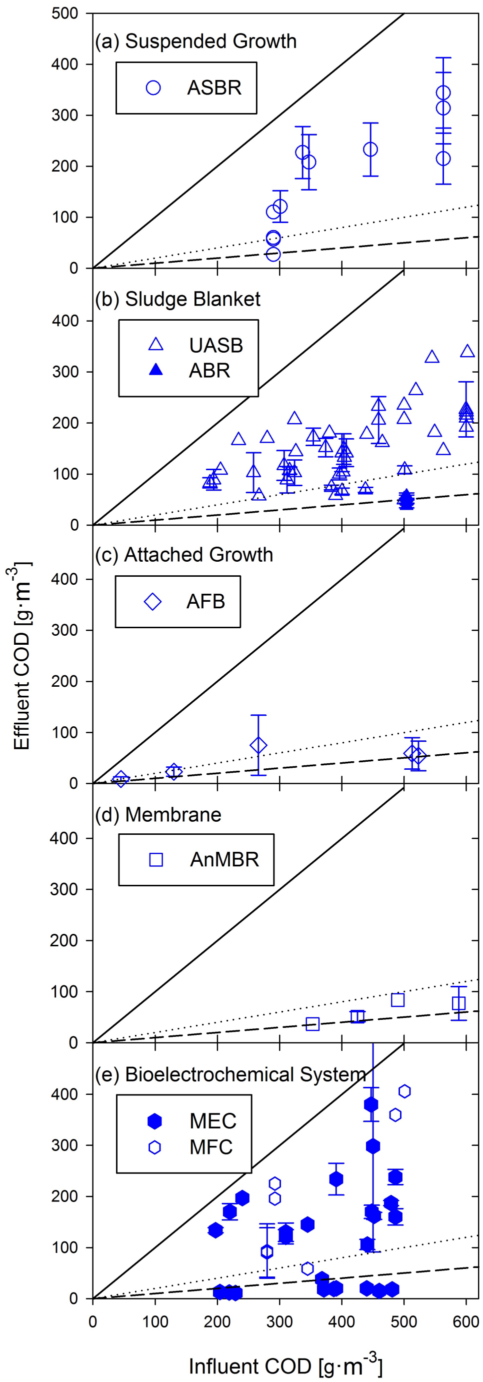


Figure 2.5: Influent vs. effluent COD [g·m$^{-3}$] from anaerobic treatment technologies treating real and synthetic wastewaters with influent COD concentrations <500 g·m$^{-3}$. Points and error bars represent averages ± standard deviations from experimental data sets. Plots (a) - (e) are separated by technology type (suspended growth, sludge blanket, etc.). The solid line is no COD removal (i.e., 0% removal), the dotted line is 80% removal, and the dashed line is 90% removal.

and biodiesel have a higher energetic potential per mass of fuel (33.2 and 37.2 MJ·kg fuel$^{-1}$ for biocrude oil or biodiesel, respectively) [67, 164] compared to direct combustion (14.2-21.4 MJ·kg fuel$^{-1}$, where the dried biomass is the fuel) [178, 179], conversion technologies suffer from low conversion efficiencies from algal biomass to fuel (HTL 25-54%; transesterification 10-30%) resulting in appreciable energetic losses as undesirable byproducts (e.g., the solid and aqueous fractions of HTL) [180]. However, liquid fuels offer distinct advantages as compared to electricity and heat (e.g., liquid fuels can be easily stored and transported), and there may be circumstances under which energetic losses are adequately offset by the convenience or life cycle impacts of liquid biofuel production. Among liquid fuels, HTL is able to yield oil from not only the neutral lipids, but also other macromolecules to achieve an appreciable increase in energy yield (Table 2.4). Given that phototrophic biomass cultivated in wastewater is often observed to have low levels of neutral lipids (~10% of dry weight) [67], HTL may have significant potential for wastewater-derived phototrophic biomass. In terms of feedstock preparation for downstream conversion, HTL and anaerobic digestion can both process biomass in a wet state [181], rather than requiring drying that can demand energy equivalent to the energetic content of the biomass itself [182]. Anaerobic digestion, in particular, is an accessible technology that is well proven at large scales (it is currently in operation at 1,300 WWTPs in the U.S. alone [158]) and has been clearly demonstrated to enable nutrient recycling to agriculture [183, 184]. A key challenge for the integration of phototrophic biomass into digestion processes, however, is maintaining an appropriate C:N ratio [185].

As expected from the consumption data, PBRs and stirred tank reactors face the greatest difficulty in becoming energy neutral or positive, while HRAPs, WSPs, and ATSs may be energy positive once harvesting and low conversion efficiencies have been overcome [186]. Although harvesting is often cited as one of the most critical energy challenges to meet [187], it is clear that mechanical mixing and aeration must also be reduced if these technologies are to be energy positive. Energy consumption for each cultivation technology (shown in Table 2.3 [kJ·m$^{-3}$] and Table A.5 [kJ·g-nutrient removed$^{-1}$]) was highest for PBRs (230-470 kJ·g-N$^{-1}$ and 1,200-2,400 kJ·g-P$^{-1}$) and stirred tank reactors (40-150 kJ·g-N$^{-1}$ and 180-730 kJ·g-P$^{-1}$), while HRAPs had relatively low energy demand (2-7 kJ·g-N$^{-1}$ and 10-60 kJ·g-P$^{-1}$) and

WSPs and ATSs required no appreciable energy input during operation. When compared to the energy yield of various conversion technologies (Table 2.4), energy production by PBRs and stirred tank reactors may have potentially favorable energy balances depending on the conversion process and variability of biomass generation within each technology. HRAPs and ATSs, however, are far more likely to be energy positive across the range of biomass yields due to minimal operational energy demands (Table 2.3). Although data was not available to estimate energy yield from WSPs, they would also have the potential to achieve energy positive treatment if energy efficient biomass harvesting can be achieved.

It is important to note that the energy consumption calculated for this study does not include the energy needed for the conversion process itself. There is a large degree of uncertainty associated with these technologies, some of which (like HTL) have yet to be implemented on a large scale for phototrophic biomass. However, there is still room for these technologies to be net energy positive when incorporating conversion energy demand. For example, PBRs obtained a maximum energy yield of ∼580 kJ·g-N$^{-1}$ for anaerobic digestion and HTL. With a cultivation energy demand of 230-470 kJ·g-N$^{-1}$ (Table A.5), there is still 110-350 kJ·g-N$^{-1}$ that can be used for driving the conversion process.

Table 2.4: Energy yield (kJ-fuel·g-nutrient removed$^{-1}$) for phototrophic cultivation technologies and select conversion processes.[a,b]

| **Technology** | **Nutrient** | **HTL** | **Anaerobic Digestion** | **Transesterification** | **Combustion** |
|---|---|---|---|---|---|
| **HRAP** | N | 75-160 | 32-160 | 34-100 | 90-130 |
| | P | 730-1,600 | 320-1,500 | 330-980 | 880-1,300 |
| **PBR** | N | 270-590 | 120-580 | 120-370 | 330-500 |
| | P | 230-500 | 100-490 | 100-310 | 280-420 |
| **Stirred Tank**[c] | P | 900-1,900 | 400-1,900 | 400-1,200 | 1,100-1,600 |
| **ATS** | N | 110-240 | 47-230 | 49-150 | 130-200 |
| | P | 580-1,300 | 250-1,200 | 260-790 | 700-1,100 |

[a] Calculations and assumptions can be found in Table A.4
[b] WSP could not be included due to lack of available biomass productivity data.
[c] Data was not available to estimate kJ-fuel·g-N removed$^{-1}$

To replace chemotrophic nutrient removal processes, phototrophic technologies must achieve efficient N and P removal below permit levels. A review of the literature revealed that the highest levels of N removal (average 78.5%) were achieved in PBRs. Although HRAPs and ATSs had similar maximum values of removal (87.2% and 90.7%, respectively, compared to 89.7% for PBR), they had higher variability in performance (Table 2.1 and Figure 2.6a). PBRs also achieved the highest consistent levels of percent P removal (Table 2.1 and Figure 2.6b). Additionally, although experimental conditions varied greatly across studies, PBRs have been demonstrated to achieve effluent concentrations below 3 g-N$\cdot$m$^{-3}$ and both PBRs and stirred tanks achieved effluent P levels below 0.3 g-P$\cdot$m$^{-3}$ (a subset of ATS studies also demonstrated effluent P concentrations below 0.3 g-P$\cdot$m$^{-3}$, but these studies had influent P concentrations below 1 g-P$\cdot$m$^{-3}$).

When compared to energy consumption data, it can be seen that the technologies that require more energy (PBRs, stirred tank reactors) tend to perform better in meeting traditional treatment objectives such as N and P removal from wastewater. They also generate more biomass and more energy per gram nutrient removed (Figure 2.3) with which to offset this energy consumption. Balancing increased nutrient removal and biomass yields (and thus, energy production) with higher energy demands will be a key challenge in the design and development of energy positive phototrophic systems.

## 2.6 Navigating a Path to Energy Positive Wastewater Management

A striking conclusion of this review was that phototrophic processes have the potential to produce 280-400% of the amount of energy as anaerobic processes on a per m$^3$ basis, given existing pollutant removal efficiencies and downstream conversion technologies. The energy recovery by anaerobic technologies reported in this manuscript (2-47%) assumes an energetic content for COD of 13.9 kJ$\cdot$g-COD$^{-1}$, which has recently been found to be a low estimation [167]. A higher energetic content would further reduce anaerobic energy recovery

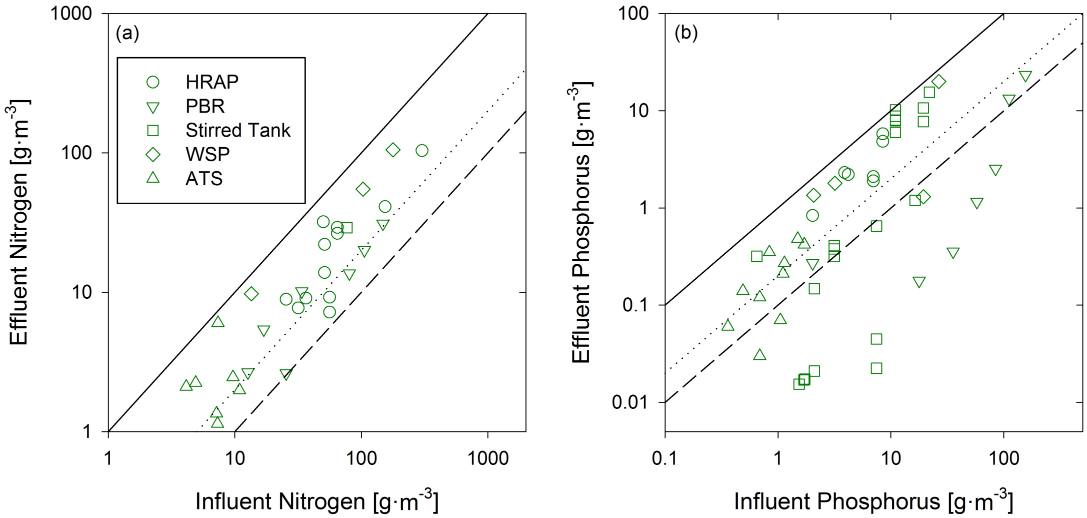


Figure 2.6: Influent vs. effluent (a) total N concentrations [g-N·m$^{-3}$] and (b) total P concentrations [g-P·m$^{-3}$] for suspended and attached growth systems on a log-log scale. The solid line identifies no nutrient removal, the dotted line 80% removal, and the dashed line 90% removal.

efficiency, whereas cultivating algae on nutrients and converting to fuels could exceed the original energetic content of the influent wastewater. Additionally, the use of nutrients for phototrophic cultivation may result in 130-510% of the energy production as would be offset by the use of nutrients for fertilization. An unfortunate finding of this review was the lack of adequate data to enable a coordinated analysis of both energy production and wastewater treatment. Of the 311 papers screened in the initial literature search, 82% could not be included because they did not measure or report adequate data. From the available data, it is clear that the potential exists for energy positive wastewater treatment and that both anaerobic and phototrophic may play a role. However, there are several critical barriers that must be overcome. Anaerobic processes must balance reduced energy consumption with increased treatment efficacy and fuel recovery, and we must develop a deeper understanding of phototrophic bioprocesses to enable process optimization. To this end, we examine the implications of this work and propose areas for future research.

### 2.6.1 Implications of This Work

This review examines the potential of various biotechnologies to directly treat domestic wastewater with a positive operating energy balance. For anaerobic technologies, influent COD is an important determinant of fuel production; higher COD concentrations lead to more energy recovery and less energy consumption (per gram of COD degraded). Since freshwater serves as a carrier for human waste in developed countries, domestic wastewater is often dilute, limiting the amount of energy that can be recovered during secondary treatment. For phototrophic technologies, a similar relationship exists between influent N and P concentrations and biomass yields. Despite limited energy recovery and production values, replacing energy intensive COD and nutrient removal processes could enable treatment plants that have already established solids digestion and on-site electricity generation to achieve energy positive operation. At the forefront of energy-conscious wastewater treatment with aerobic COD removal and BNR processes, an activated sludge WWTP in Strass, Austria has achieved energy self-sufficiency by implementing a high rate aerobic process, anammox treatment of nutrient rich side streams, and on-site electricity generation from biogas generated by solids digestion. A published COD mass balance and energy analysis of the plant indicated that 75% of the COD entering the plant is fed to a digester (61% primary and high rate solids and 14% waste solids from biological nutrient removal) and 36% is converted to biogas [188]. The aerobic BNR process, in which 31% of the influent COD and 80% of the N is removed, accounted for 45% of energy consumption at the plant. The Strass WWTP COD mass balance was used to simulate the energetic potential of replacing the existing aerobic processes with anaerobic and phototrophic wastewater treatment. If the BNR process was replaced with an ABR to remove COD and a HRAP to remove nitrogen, total plant biogas production could potentially increase by 39% and energy recovery from COD could reach 41% (Section A.5). The energetic content of biomass produced in the HRAP during N removal (2,200 kJ$\cdot$capita$^{-1}\cdot$day$^{-1}$) was estimated to be more than twice as much as recovered biogas (1,020 kJ$\cdot$capita$^{-1}\cdot$day$^{-1}$). If PBRs were employed rather than HRAPs, the estimated biomass energy content alone (7,800 kJ$\cdot$capita$^{-1}\cdot$day$^{-1}$, assuming N-limited growth) could be more than three times the caloric energy content of wastewater entering the plant (2,500 kJ·capita$^{-1}$·day$^{-1}$). More broadly, combined anaerobic and phototrophic processes could reduce energy demand and achieve energy recovery and production on the order of 5.0-9.2 kWh·m$^{-3}$ (using higher values for UASBs and PBRs) - well above the whole-plant energy demand of conventional WWTPs (0.3-0.6 [55, 56]).

Though achieving energy and resource positive treatment in developed countries is an important goal for future treatment, far more urgent is the need to deploy sanitation infrastructure in developing and underdeveloped communities, where an estimated 2.5 billion people lack access to improved sanitation [189]. Even in cases where individuals have access to bathroom facilities and collection systems, it is estimated that globally 1.5 billion people connected to sewerage infrastructure have their wastewater discharged without treatment [190]. In developing communities in tropical regions, mainstream anaerobic treatment of domestic wastewater has been shown to be a viable means of achieving treatment goals while simultaneously producing biogas [191]. This biogas, if utilized properly, could be an invaluable resource providing a consistent supply of electricity. In developing countries, effluent from anaerobic treatment processes (e.g., UASB) can be fed to WSP for further treatment [192, 193]. The data analysis presented in this review indicates that converting WSPs to HRAPs is a path toward more meaningful energy production from wastewater management. Ultimately, one of the greatest opportunities to advance wastewater treatment in developing communities is to recover resources that make wastewater management energy positive and financially viable.

### 2.6.2 Future Research Needs - Anaerobic Technologies

The experimental results complied in this review clearly show that energy recovery in the form of methane gas is significantly higher than energy recovery by MECs and MFCs. While methane-producing technologies do not require electrodes or applied voltage to generate fuel, converting biogas to electricity requires expensive auxiliary equipment (i.e., gas conditioning, storage, prime movers or fuel cells) and is currently only feasible at high flow wastewater treatment facilities (>30 MGD) [158]. Of the more the 1,300 treatment plants that employ anaerobic digestion for solids management in the U.S., only 364 are sites generate enough biogas to make combined heat and power (CHP) financially viable, of which 104 currently generate electricity from biogas [158]. Primary anaerobic treatment would make CHP accessible to smaller WWTPs, but it remains to be seen at what scale economic feasibility could be reached.

Though methane is relatively insoluble in water (Henry's Constant = 776 bar·L·mol$^{-1}$), loss of dissolved methane in the wastewater effluent continues to be a critical challenge for anaerobic processes [194, 195]. This loss of fuel removes much of the potential for anaerobic processes to be energy positive, especially since energy savings from psychrophilic operation are in tension with increased energy losses due to higher methane solubility at lower temperatures [33]. Finding alternative methods to recover dissolved methane without excessive energy input (i.e., using an amount of energy greater than the amount recovered) will be pivotal to achieve energy positive treatment with AnMBR.

In terms of energy recovery, MFC bioelectricity was significantly lower than gaseous products. However, when fuel conversion to electricity was considered, the discrepancy between MFCs and gas-producing technologies was less substantial, indicating that MFCs may be a favorable option for distributed electricity production from wastewater. To capitalize on this potential, research efforts should focus on anode and passive-air cathode fabrication without the use of expensive metals as well as evaluation of power production from source separated waste streams. METs can also be designed to operate in concert with methane-producing processes to enhance treatment efficiency and recover nutrients. Allocating a portion of soluble organic energy to produce electrical current with MET electrodes could be leveraged toward electrolytic pH adjustment to volatilize and concentrate ammonia [196–198] or recover N and P as struvite [199, 200]. Ionic current produced by MET could also be used to polarize capacitor electrodes and remove charge molecules such as nutrients and minerals from wastewater [201].

### 2.6.3 Future Research Needs - Phototrophic Technologies

Although the predominant focus of nutrient research in the wastewater field has been on improving the efficiency of BNR by chemotrophic bacteria, the energetic potential of phototrophic processes warrants further development of these processes for energy positive nutrient management. In particular, more highly engineered systems that minimize footprint (like PBRs and stirred tank reactors) may have potential in advancing nutrient removal initiatives while also increasing the energy independence of treatment facilities. A critical challenge in achieving reliable and resilient phototrophic treatment systems, however, is a lack of understanding of how process design and operational decisions influence effluent quality, biomass productivity, and biochemical composition [169]. Developing a deeper understanding of mixed community phototrophic biotechnology in the context of wastewater treatment will require long-term experimentation with real wastewaters under natural light (or simulated natural light) conditions with diurnal cycles. Targeted experimentation and modeling may enable process optimization, but a priority should be to determine how complex models will need to be to enable reliable predictions of performance across climates and wastewaters [202, 203].

Harvesting and downstream processing to usable fuels are also opportunities for technology advancement, including research furthering the development of processing technologies that do not require complete drying of biomass prior to processing: anaerobic digestion and HTL hold particularly high potential in this regard. In addition to fundamental advancements to HTL and the management of waste products [204], a critical challenge is to link process design decisions with downstream processing to usable energy. Without a mechanistic understanding of the links among cultivation decisions, biochemical composition, harvesting, and processing to fuel, any attempts at process optimization are likely to result in tradeoffs that may obscure energetic impacts of design and operational modifications.

### 2.6.4 Conclusion

The pursuit of energy positive domestic wastewater treatment is a necessity due to both the financial costs and the broader environmental impacts incurred by energy consumption. Beyond economic and environmental drawbacks, energy intensive treatment processes are also a financial burden for developing communities that may even lack the energy infrastructure to reliably treat wastewater aerobically. Based on the results of this review, it is clear that WWTPs can be net energy producers, especially if phototrophic technologies are leveraged to increase the energetic potential of wastewater through inorganic carbon fixation. In the search for energetically favorable technologies, however, there is a critical point to be made: we should not compromise traditional sanitary engineering objectives for wastewater treatment systems (i.e., effluent quality) to achieve energy positive performance, but rather seek to develop technologies that achieve equivalent or superior effluent quality by leveraging biological, chemical, and physical processes whose treatment efficacy is not in direct tension with their energy balance. Therefore, we should seek to advance technologies that have synergies between effluent quality and energy production, such as anaerobic and phototrophic technologies where every gram of pollutant removed increases the potential energy yield from the system.

# CHAPTER 3

# ADVANCING ANMBR DESIGN BY IDENTIFYING A PATHWAY FOR SUSTAINABLE TECHNOLOGY INNOVATION

## 3.1 Introduction

People are becoming increasingly aware that their actions have broader environmental impacts and therefore are making a concerted effort to quantify them. As a result, there has been a movement toward considering upstream and downstream impacts when planning a project (i.e., before construction begins and after the useful life has ended, respectively). However, this is particularly challenging for certain aspects of society, such as the wastewater treatment industry. The primary goal when designing a wastewater treatment plant (WWTP) is to protect public and environmental health, which has traditionally been accomplished using aerobic treatment processes [26]. Aerobic treatment has been around for over a century because it consistently removes contaminants, but consumes a great deal of energy in the process. In the U.S., municipal WWTPs account for ∼3% of the national yearly energy demand (0.3-0.6 $kWh \cdot m^{-3}$ of wastewater treated) [14, 55, 56], approximately 50-60% of which is due to aeration [205]. While this energy demand does result in high electrical bills for the WWTPs, it also negatively impacts the environment (e.g., $CO_2$ release contributes to global warming potential). Only recently have wastewater engineers begun trying to account for broader impacts that result from different stages in a wastewater treatment plant's life cycle [5]. Given the high energy consumption of aerobic processes, there is a need to examine alternative technologies that have the potential to reduce cost and environmental impacts while still holding human health paramount.

Anaerobic wastewater treatment has been shown to have several benefits over traditional aerobic processes [19]. Energy consumption is lower primarily because aeration is not required [26]. Additionally, methane is produced, which can be used to generate electricity.

Microbes grow more slowly because less energy is available as a result of the biochemical reactions that govern this process; because of this, less sludge needs to be wasted [25]. However, anaerobic processes do have some drawbacks. While $CH_4$ can be used for electricity generation, recovering 100% of the produced biogas is difficult [29]. Given that $CH_4$ is around 28 times worse than $CO_2$ as a greenhouse gas (GHG) if the unrecovered methane is released to the atmosphere [2], the WWTP would have an even greater negative impact on the atmosphere. Though slow-growing organisms result in less sludge, they also require a longer solids retention time (SRT), which can prove operationally difficult [206]. The principal drawback, however, is that anaerobic treatment is at times unable to meet discharge requirements, especially in the case of COD removal [19]. This hurdle will have to be overcome in order to ensure that the environment, and more importantly society, remain healthy.

Anaerobic membrane bioreactors (AnMBRs) have some of the benefits of other anaerobic processes (e.g., methane production and lower sludge wasting rates) while also producing a consistently high quality effluent [19]. They combine a wastewater treatment process [e.g. completely stirred tank reactor (CSTR) or anaerobic filter (AF)] with membrane filtration [30] . The inclusion of membranes in the wastewater treatment process allows hydraulic retention time (HRT) to be decoupled from SRT; this often results in much smaller plant footprints, which can lower capital costs [31]. However, more energy is consumed with these processes (compared to other anaerobic WWTP designs) due in large part to membrane fouling control (e.g. gas sparging) [183]. AnMBRs are also a relatively new technology, so a multitude of treatment designs have been examined in order to find a layout that could be used as a full-scale plant [28, 32–34, 37–44]. With all these available configurations, some are bound to be inherently better than others in terms of cost and environmental impact.

In order to elucidate AnMBR designs that have greatest potential to be utilized at a full-scale plant in the future, all the different configurations must be compared in a standardized manner so as not to introduce any bias. Assuming that with additional research and optimization a given layout is able to meet discharge limits, deciding factors then become cost and environmental impacts. Following the general methodology discussed in [16], the objective of this work was to utilize a life cycle assessment (LCA) and life cycle costing (LCC) framework to compare different designs objectively and ascertain competitive advantages or limiting pitfalls for each design. To that end, a model was built in MATLAB to navigate the decision space for these designs (i.e., the potential operating conditions that a plant could experience at full scale). Additionally, uncertainty and sensitivity analyses were conducted on the input parameters in order to determine which need to be examined in greater detail going forward as well as which have the greatest impact on the model outputs.

## 3.2 Methods

### 3.2.1 Goal and Scope Definition

For this study, the methodology of ISO 14040 was utilized to compare environmental impacts of different AnMBR designs [207]. A functional unit of 1 $m^3$ wastewater (400 mg COD$\cdot L^{-1}$) treated to discharge quality (30 mg COD$\cdot L^{-1}$) was used. The life span of the treatment plant was assumed to be 30 years. The system boundary included both construction and operation of the plant; demolition was excluded (Figure B.1, consistent with [208–211]). First and second order environmental impacts were also examined (i.e., direct emissions from WWTPs and emissions from upstream electricity and material production, respectively) as well as avoided energy production impacts from biogas recovery and utilization.

### 3.2.2 Life Cycle Inventory (LCI)

The LCI focused on materials needed to construct and operate the plant (e.g., concrete) as well as more generic materials (e.g., reinforcing steel for the concrete) and the transportation costs of these materials. For construction, the inventory included concrete for the reactors and pump/blower buildings, volume of excavation, piping (stainless steel), membrane material, and a combined heat and power (CHP) system.

A multiplicative approach was utilized to account for WWTP construction materials as outlined by [212] and used by [213]. Transportation by rail or lorry and electricity consumption were also included. For operation, considered processes included citric acid and NaOCl consumption for membrane cleaning, electricity consumption and offsets, granular activated carbon (GAC) replacement, membrane replacement, and sludge landfilling. Air and water emissions from the plant were also included in this study (i.e., COD, $NH_3$, $NH_4^+$, organic nitrogen, phosphorus, and $PO_4^-$ for water; $CH_4$ and $CO_2$ for air). However, LCI items must be expressed as unit processes - the smallest elements of life cycle inventory data - to be used in following steps [214]. Ecoinvent v3.0 was used to convert the LCI to unit processes [215].

### 3.2.3 Life Cycle Impact Assessment (LCIA)

Based on the LCI results, the LCIA was conducted using the Tool for Reduction and Assessment of Chemical & Other Environmental Impacts (TRACI V2.0), which characterizes the impact each item in the LCI has on nine categories: ozone depletion, global warming potential (GWP), smog pollution, acidification, eutrophication, carcinogens, non-carcinogens, respiratory illness, and ecotoxicity [216]. Additionally, electricity used to run the plant was assumed to be generated from several sources based on the state of Illinois [217], namely: 47.8% nuclear, 46.5% hard coal, 2.9% natural gas, 2.6% renewables, and 0.1% from both hydroelectric and oil.

### 3.2.4 Life Cycle Costing (LCC)

Using the system boundary described above, life cycle costs for the construction and operation unit processes were calculated using CAPDETWorks (by Hydromantis, Inc.) and equations provided by Hazen and Sawyer Engineers. While construction incurs a one-time capital cost, operation and maintenance (O&M) results in yearly costs. In order to compare the costs of different configurations, the net present value of the O&M costs was calculated using an interest rate that varied between 6-10% uniformly. Two CAPDETWorks equations were discontinuous and so were adjusted in order to allow costs to scale proportionally with variations in input parameters (Table B.3). Ranges over which the firm pumping capacity equations were applicable for pumping were changed according to where the two equations intersected. Additionally, the equation used to calculate the installed pumping equipment cost was replaced with a polynomial equation fit to the CAPDETWorks costs calculated over a range of flow rates. Equipment costs were provided by CAPDETWorks or were retrieved directly from the manufacturer.

### 3.2.5 Design Roadmap

When selecting a WWTP configuration, many design decisions must be made, such as which reactor type to use or what membrane type to utilize. To simplify the design process while also examining a sufficiently large range of AnMBR configurations, the following discrete decisions were considered: reactor type, reactor configuration, membrane type, membrane material, operation mode, physical cleaning method, chemical cleaning method, soluble methane management, and methane processing method (Figure 3.1). Some of the initial decisions decrease the number of choices that are available in future decisions; these limitations were determined during conversations with Hazen and Sawyer and are detailed below. Ultimately, the overall costs and environmental impacts of 224 different AnMBR designs were compared.

#### Reactor Type

The two reactor types considered in this study were the CSTR [33, 34, 37, 42] and AF [218]. For the CSTR, the volume of the reactor predominantly depends on the HRT. The volume of the AF is determined by the organic loading rate (OLR) of the influent stream because this influences the volume of packing media required. This decision also considered the inclusion of a down-flow aerobic sponge filter (AeF) [219] or GAC [28, 39, 44] in the system. The AeF can be paired with the AF and the size of the reactor depends on the OLR of the stream exiting the AF. Both the CSTR and AF are able to utilize GAC, which is dosed into the reactor depending on the GAC concentration and reactor volume being used.

#### Reactor Configuration

Two membrane configurations that were examined in this study: submerged [38] and cross-flow [32, 42]. While CSTRs can utilize either configuration, AFs are not compatible with

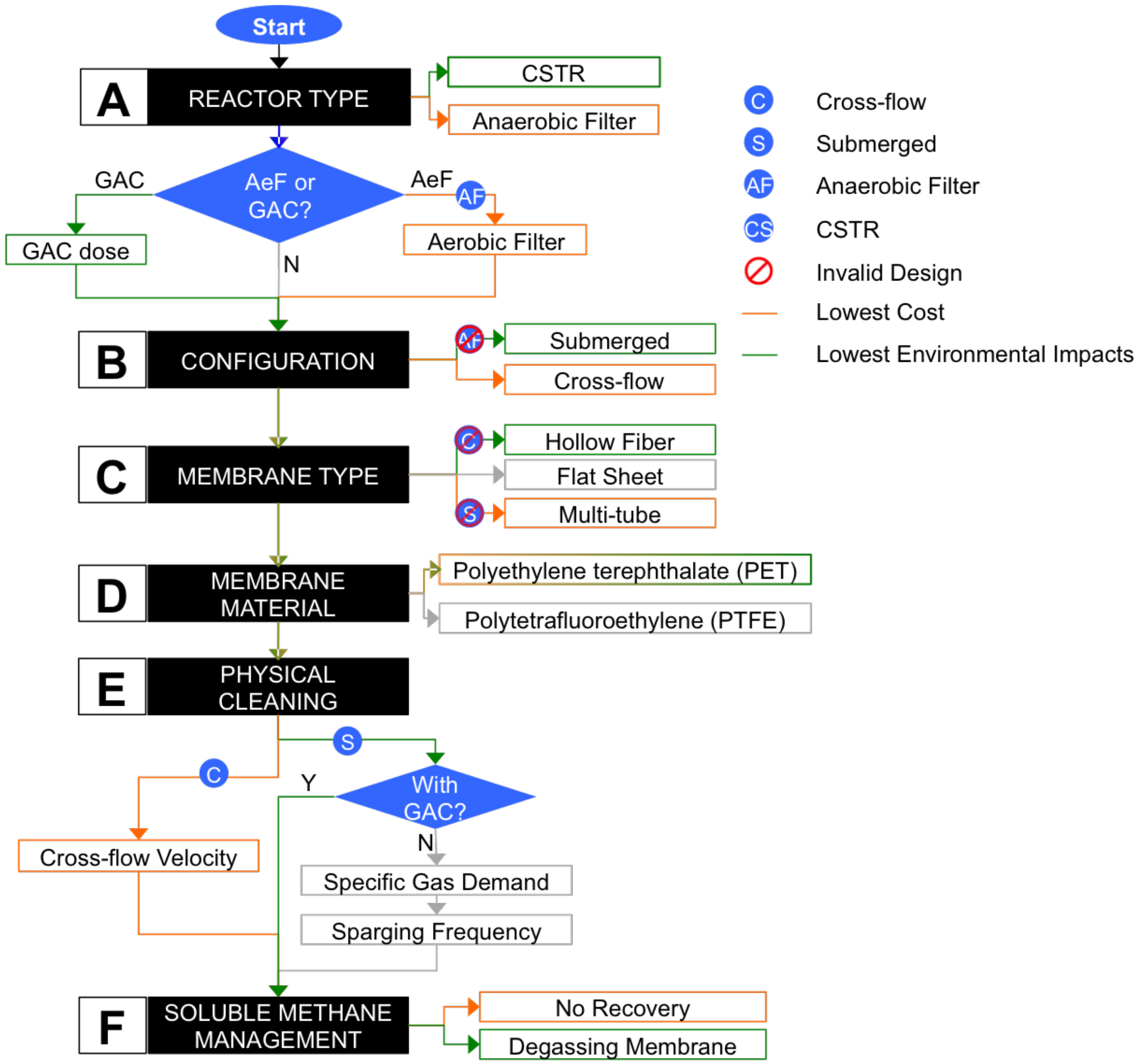


Figure 3.1: Roadmap detailing the different decisions to consider when designing an AnMBR. Choices that are dependent on preceding decisions are indicated by blue circles. Paths are colored to indicate which designs had the lowest cost or environmental impacts. Two-color boxes indicate designs that utilized the same decision. Grayed paths were not utilized by either of the optimal designs.

submerged membranes.

### Membrane Type

The three membrane types considered in this study are hollow fiber (HF) [28, 37, 38, 43, 44], flat sheet (FS) [33, 41, 42], and multi-tube (MT) [218, 220]. While FS membranes can be used in both CSTRs and AFs, HFs are usually only used in submerged AnMBRs. Similarly, MT membranes are typically seen in cross-flow configurations. In order to include specific membrane information (e.g., nominal surface area per module) a default membrane was assumed for each type: GE ZeeWeed* 500D for HFs, Kubota RM515 for FS, and Pentair X-flow for MT. The assumed useful life of all membranes was 7 years, as suggested by Hazen and Sawyer.

### Membrane Material

Two membrane materials were included in the LCA analysis: polyethylene terephthalate (PET) and polytetrafluoroethylene (PTFE). These two materials were chosen because they are commonly used for membrane bioreactors and because they could be accounted for using TRACI. While these materials incur different environmental impacts due to their respective production methods, differences in costs were assumed to be negligible.

### Operation Mode

AnMBRs are often operated as filtration-relaxation followed by periodic backwashing. A common backwashing duration ($t_b w$) is 30 seconds for every half-hour of operation (0.4 hr·day$^{-1}$). An AnMBR system can then be divided into $24/t_b w$ units, with only one unit backwashing at a given time. In this instance, the available membrane area must then be increased in order to compensate for the membrane being backwashed. For this study, the assumed backwashing time was 0.4 hr·day$^{-1}$ [220].

### Physical Cleaning

Physical cleaning is necessary in order to remove foulants and varies depending on the reactor configuration, influent wastewater, and operational conditions [220]. Cross-flow systems utilize a high cross-flow velocity to provide foulant removal. Submerged AnMBRs are commonly cleaned using gas sparging. However, if a submerged system also contains GAC, the granules have been shown to clean the membrane during recirculation flow, negating the gas sparging requirement [28].

### Chemical Cleaning

For further foulant removal, chemical cleaning both in-place (CIP) and out-of-place (COP) is required. Citric acid (100% by weight) and sodium hypochlorite (NaOCl, 12.5% by weight) were included in this study for inorganic and organic foulant removal, respectively. Annual consumptions were assumed to be 600 gal$\cdot$yr$^{-1}\cdot$MGD$^{-1}$ and 2,200 gal$\cdot$yr$^{-1}\cdot$MGD$^{-1}$ for citric acid and NaOCl, respectively, as suggested by Hazen and Sawyer.

### Soluble Methane Management

AnMBR operation at ambient temperatures has been shown to result in an effluent stream that contains upwards of 50% of the produced methane dissolved in water [19]. Therefore, including a degassing membrane (DM) in the design was considered in order to quantify the tradeoffs between cost and environmental impact of releasing the methane to the atmosphere [194].

### Methane Processing

Biogas produced and collected during the anaerobic process was assumed to be reused for energy and heat generation using a combined heat and power (CHP) system. Four CHP systems were considered in this study: internal combustion, combustion gas, microturbine, and fuel cell; their associated efficiencies can be found in Table B.1 [158, 221].

### 3.2.6 Uncertainty and Sensitivity

Monte Carlo with Latin Hypercube Sampling (LHS, 1,000 trials per AnMBR design) was used to propagate input uncertainty for 19 parameters (values in Table B.2) to costs and life cycle environmental impacts for each design. Assigned values were based on values reported in literature or were conservatively estimated if data were lacking. Uncertainty distributions were also assigned to each parameter based on data availability; uniform distributions (i.e., $\pm$ 20% of the assigned value) were used unless evidence suggested otherwise. A sensitivity analysis was also performed in order to elucidate which inputs had the greatest impact on environmental impacts and costs.

## 3.3 Results and Discussion

### 3.3.1 Influence of Design Decisions on Performance

During examination of the different AnMBR designs, variations in the results due to different CHP plant types was found to be relatively insignificant (i.e., <5% of the total cost and environmental impacts) and were therefore excluded from further analysis, reducing the number of designs to 56. Many AnMBR designs were found to have some beneficial aspects whether in terms of LCC or LCA results. In order to examine the feasible designs objectively, the net present value of the life cycle cost was plotted against the nine environmental indicators (Figure B.2). In general, designs that were identical save for the decision whether or not to include a DM had similar results; those with a DM had a higher total cost, but lower environmental impacts.

The degassing membrane is an early-stage technology that requires a great deal of power to operate (0.042 $kW{\cdot}m^3$ [194]). This additional electricity consumption adds to the cost and environmental impacts of a given design. However, assuming the process is 100% efficient and that 30-50% of methane is lost in the effluent [28, 29], including a DM results in a net decrease in environmental impacts because (1) methane is not emitted to the atmosphere and (2) this additional methane can be used to generate more electricity. Though Bandara et al., 2011 did not operate a full-scale plant when examining the use of a DM, this technology shows promise for preventing fugitive methane release.

Within these plots, there are distinct areas where points aggregate as a result of one or two specific design decisions. For example, looking specifically at LCC vs. GWP, the highest costs are due to gas sparging (i.e., physical cleaning for a submerged membrane without GAC or an AeF) (Figure 3.2, red shaded area). Apart from the effects of gas sparging, the cross-flow FS configuration had the highest costs (blue shading). Including GAC in a CSTR resulted in elevated costs, but lower GWP (orange shading) compared to the designs that had the lowest cost (i.e., cross-flow MT configurations, green shading). Similar trends exist with all the environmental impact factors examined, indicating that gas sparging (due to a submerged membrane) or a cross-flow configuration with flat sheets incur higher costs or impacts and should therefore be avoided when designing an AnMBR.

While including granular activated carbon does incur more costs, many designs (i.e., 60-100% of the lowest five emitters for each environmental impact category) that resulted in low emissions contained GAC. This was because the GAC aids in membrane fouling prevention [28], which was assumed to replace the need for other physical cleaning (e.g., gas sparging), ultimately reducing electricity consumption. Based on this, including GAC in AnMBRs could prove to help remove some of the barriers related to full-scale implementation.

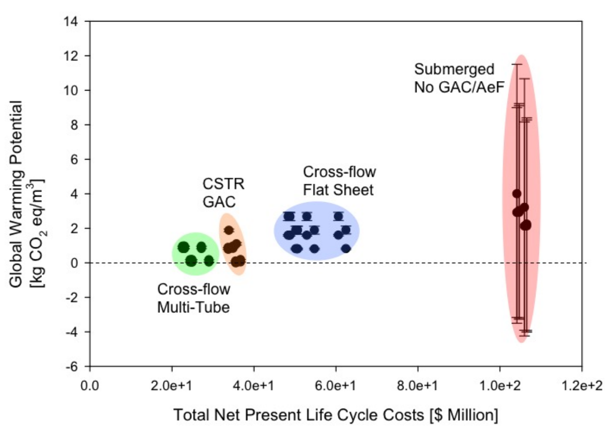


Figure 3.2: Average net present life cycle cost [$] plotted against global warming potential [kg $CO_2$ eq.$\cdot$m$^{-3}$]. Error bars indicate standard deviation across 1,000 trials. Shaded areas indicate predominant characteristic(s) affecting cost and/or GWP.

### 3.3.2 Impact of Construction vs. Operation and Maintenance on Results

The two main stages of the AnMBR's life cycle included in this study were construction and O&M. In order to determine which stage had a greater impact on the total for a given metric, the percent contribution for each stage was calculated during each trial and was then averaged for each design (Figure B.3). Averages across the 56 designs were then determined in order to examine the overall effect of these two stages on LCC and LCA results (Figure 3.3).

On average, construction accounted for 20% of the net present cost for any given design indicating that operation and maintenance has a large impact on the overall cost of a project. In terms of environmental impacts, construction had a greater impact (i.e., >50% of the total) on ozone depletion, carcinogens, non-carcinogens, and ecotoxicity (Figure 3.3). Operation and maintenance was the predominant contributor for GWP, smog, acidification, eutrophication, and respiratory disease.

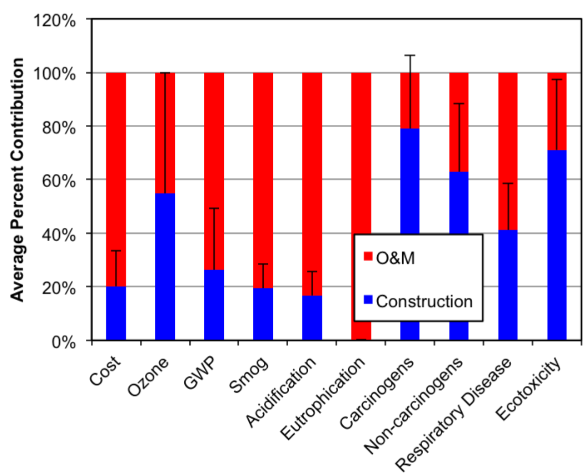


Figure 3.3: Average percent contribution of construction and O&M to total net present life cycle cost and environmental impact across 56 AnMBR designs (1,000 trials per AnMBR design, error bars indicate standard deviation).

### 3.3.3 Uncertainty and Sensitivity Analysis Results

Variations in cost and GWP resulting from the six design decisions examined in this study were quantified by normalizing the maximum and minimum difference between two configurations that only varied by one decision (e.g., the AnMBRs are the same, but one uses a submerged membrane and the other has a crossflow membrane) to the overall range (i.e., across all 56 configurations, Figure B.4). For cost, deciding whether to include additional processing (i.e., GAC or AeF) and membrane configuration had the greatest potential variation (85 and 66%, respectively). Additional processing and membrane type resulted in the largest variation for GWP (53 and 43%, respectively). Additional processing affected the outputs to such an extent because some of the configurations that excluded GAC or an AeF required gas sparging, greatly increasing electricity demand and therefore cost and GWP.

The sensitivity of total net present life cycle cost [$] and GWP [kg $CO_2$ eq$\cdot$m$^3$treated$^{-1}$] to the 19 input parameters across all configurations was determined by comparing Spearman's rank correlation coefficients [222] (Figure 3.4). For cost, the most sensitive input was flow rate, with a median correlation coefficient above 0.4 (i.e., a moderate correlation), which was significantly different from the hydraulic loading for the AeF (p-values $<0.001$, $\alpha=0.05$; two-tailed, unpaired t-test). Hydraulic loading was used for comparison because it had very little effect on cost (median correlation coefficient of 0.0009). Flux and interest also had elevated medians (i.e., $<-0.1$), indicating a weak negative correlation.

For GWP, the most sensitive input was flux, which was shown to have a moderate negative correlation ($\sim$-0.4). This was statistically different from interest, which has no effect on environmental impacts (p-value $<0.001$). Though they had medians close to zero, cross-flow velocity and specific gas demand had high variability across the decision space (i.e., 90th percentiles above 0.3 and 0.5 for cost and GWP, respectively), indicating that some designs are more sensitive to these parameters.

### 3.3.4 Some Designs Possess Competitive Advantages

Of the configurations considered, two were found to be the most promising in terms of either costs or environmental impacts (Figure 3.1 and Table 3.1). The AF that included an AeF as well as a cross-flow MT membrane made of PET had the lowest cost (orange path in Figure 3.1). The CSTR with GAC and a submerged HF membrane made of PET had the lowest environmental impacts (green path). Of particular interest is that the design with the

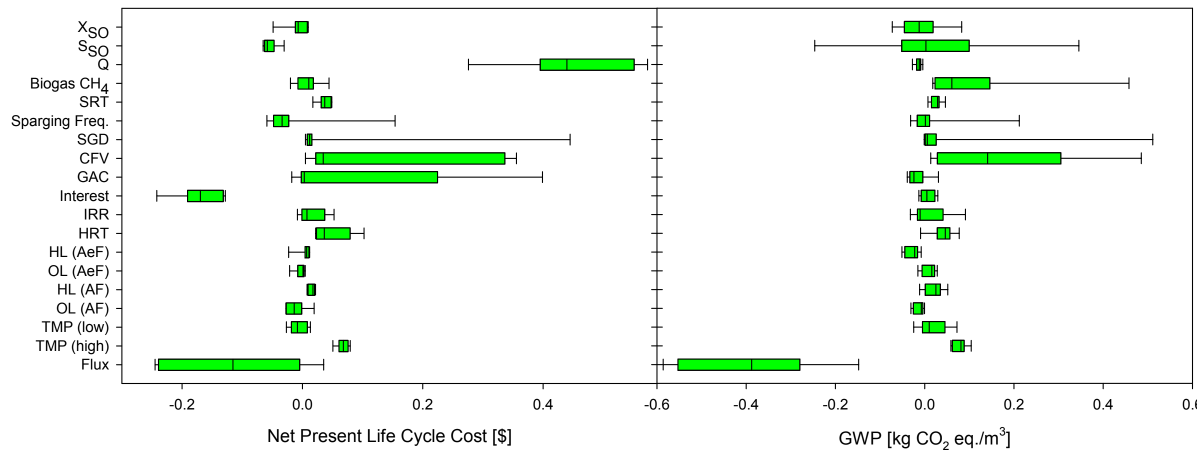


Figure 3.4: Sensitivity of total net present life cycle cost [$] and GWP [kg $CO_2$ eq$\cdot$m$^3$treated$^{-1}$] to 19 input parameters. Values were determined using the results from Spearman's rank correlation calculations for all 56 configurations. Negative values indicate an inverse correlation between the input parameter and the output value. Tails are the 10th and 90th percentiles.

lowest environmental impacts was shown to remove pollutants from the environment (i.e., GWP, smog, and acidification), indicating that this design has the potential to reverse the effects of emissions from other WWTPs. While carbon trading was outside the scope of this study, being able to sell carbon credits could potentially offset the additional costs incurred by including GAC and a DM system.

Examining the two designs in terms of cost, the greatest contributors to yearly O&M costs are electricity and membrane replacement (approximately $350,000 and $1,000,000 yearly, Figure B.5). Taking into consideration a 30-year useful life, these costs are the primary reason O&M contributes 80% of the total net present cost on average. Focusing on environmental impacts, the lowest cost configuration is dominated by steel production (for construction), pumping processes (namely permeate, lift, and retentate pumping), and miscellaneous operation (for O&M, Figure B.6). The design that had the lowest environmental impacts was also influenced by steel (though to a lesser extent, reflecting the lower steel requirement), but was affected more by miscellaneous construction (e.g., GAC and DM impacts) and concrete. While the construction impacts were higher for this design, decreases in O&M impacts overcame higher construction impacts, due mainly to decreases in permeate pumping, recirculation pumping, and miscellaneous operation.

Looking specifically at these two designs, it is clear which inputs each of these designs is most sensitive to (Figure B.7). For the configuration with the lowest environmental impacts, cost is most sensitive to flow rate and GAC dose (correlation coefficients of 0.46 and 0.40, respectively). GWP for this configuration is most sensitive to the influent soluble substrate concentration (-0.45). The total cost of the configuration with the lowest cost was most sensitive to flow rate (0.57). GWP of this configuration was influenced mostly by fugitive $CH_4$ and the influent soluble substrate concentration (0.46 and 0.35, respectively). It is interesting to note that while the GWP of both configurations are sensitive to influent substrate, this parameter has opposite effects on the results. This is likely due to the effect of substrate on the biogas production rate. For a design that is able to recover methane well, high biogas productivity will not have any effect on the environment. However, if the design loses a lot of methane to fugitive emissions, greater methane production only serves to exacerbate the problem.

Table 3.1: Average total ± standard deviation of net present cost and environmental impacts for the two designs that had the lowest value for at least one category (lowest values in bold). Units for environmental impacts are kg equivalents per $m^3$ treated.

| | AF AeF Crossflow MT PET no DM | CSTR GAC Submerged HF PET DM |
|---|---|---|
| Total Net Present Cost [$] | **$22.6 ± 5.6 M** | $35.5 ± 11.4 M |
| Ozone [CFC-11] | $7.09 \times 10^{-9} \pm 1.52 \times 10^{-35}$ | $\mathbf{2.74 \times 10^{-9} \pm 7.42 \times 10^{-36}}$ |
| Global Warming Potential [$CO_2$] | $8.29 \times 10^{-1} \pm 9.37 \times 10^{-5}$ | $\mathbf{-1.03 \times 10^{-3} \pm 3.28 \times 10^{-8}}$ |
| Smog [$NO_x$] | $5.37 \times 10^{-3} \pm 4.00 \times 10^{-12}$ | $\mathbf{-1.37 \times 10^{-4} \pm 1.76 \times 10^{-13}}$ |
| Acidification [$H^+$] | $2.67 \times 10^{-2} \pm 4.66 \times 10^{-9}$ | $\mathbf{-2.33 \times 10^{-3} \pm 2.77 \times 10^{-10}}$ |
| Eutrophication [N] | $2.82 \times 10^{-2} \pm 3.78 \times 10^{-20}$ | $\mathbf{2.81 \times 10^{-2} \pm 2.81 \times 10^{-21}}$ |
| Carcinogens [Benzene] | $1.01 \times 10^{-8} \pm 6.33 \times 10^{-34}$ | $\mathbf{7.19 \times 10^{-10} \pm 8.70 \times 10^{-41}}$ |
| Non-carcinogens [Toluene] | $1.42 \times 10^{-8} \pm 1.61 \times 10^{-33}$ | $\mathbf{1.57 \times 10^{-9} \pm 2.54 \times 10^{-38}}$ |
| Respiratory [PM 2.5] | $1.86 \times 10^{-4} \pm 1.75 \times 10^{-17}$ | $\mathbf{1.96 \times 10^{-5} \pm 3.19 \times 10^{-20}}$ |
| Ecotoxicity [2,4-dichlorobenzene] | $1.41 \times 10^{-1} \pm 1.19 \times 10^{-5}$ | $\mathbf{2.41 \times 10^{-2} \pm 1.07 \times 10^{-11}}$ |

### 3.3.5 A Path Forward for AnMBR Research and Design

While AnMBRs are an emerging technology, they have been shown to be able to achieve reliable removal of COD from municipal wastewater [33,223]. As further research is conducted on AnMBRs, treatment efficacy is expected to improve in conjuction with decreases in life cycle costs [224]. Though not all of these designs have been experimentally tested, identifying which technology development pathways have the greatest potential to be financially viable and have low life cycle environmental impacts will serve to accelerate sustainable wastewater engineering practices.

Though this novel analysis of AnMBR designs robustly examined many configurations, there were some design aspects that should be included in future work in order to represent a full-scale municipal WWTP better. For example, membranes can be made out of many different materials, including other types of plastic, ceramic, and sintered steel. Each of these is produced differently and therefore results in varying costs and environmental impacts. Foam and odor control, while not directly involved in treating wastewater, may also be required for full-scale operation. A comparison of the results of this study to actual plants (whether lab- or full-scale) could also show a compelling dichotomy of how AnMBRs are currently designed versus how they should be configured going forward.

The results of this study showed that certain designs have the capability of carbon-negative performance as well as smog and acidification reduction in the environment. While there were no designs that had both minimum costs and minimum environmental impacts, this study showed that certain designs should be avoided all together due to their significantly higher results (namely designs that require gas sparging or those that use a cross-flow flat sheet configuration). If future research focuses on the designs that were identified in this study as having a competitive advantage, net positive energy production and negative environmental impacts can be achieved. In the U.S., WWTPs need to be upgraded in the coming decades to meet more stringent discharge requirements as well as update worn-out infrastructure. Utilizing AnMBRs instead of energy-intensive aerobic processes could prove to be both an economic and environmental asset.

# CHAPTER 4

# CONCLUSIONS AND ENGINEERING SIGNIFICANCE

## 4.1 Overview

The overarching goal of this thesis is to accelerate the pursuit of sustainability in wastewater treatment; a particular challenge given the charge of this field to protect the health of society and the environment. This work began by introducing a multitude of alternative wastewater treatment technologies (anaerobic and phototrophic) with the primary goal of characterizing demonstrated treatment performance and identifying critical barriers to energy positive wastewater management (Chapter 2). In doing so, operational and performance targets were determined that would enable those technologies with the greatest potential for energy positive carbon, nitrogen, and phosphorus management to succeed at a full-scale treatment plant. AnMBRs were shown to have consistently high COD removal, but a key challenge facing this technology is the energy consumption due to fouling prevention. However, not all AnMBR designs require energy-intensive cleaning measures. In an effort to compare different AnMBR configurations without bias, a QSD framework was implemented to elucidate the competitive advantages or serious pitfalls under the assumption that all designs will one day be able to treat an influent wastewater to a uniform effluent quality (Chapter 3). There was no one design that had the lowest costs or environmental impacts, rather, the results served to reinforce the need to examine tradeoffs between economic and environmental factors.

## 4.2 The Potential of Emerging Technologies

A novel aspect of this work is the comparison of existing, published anaerobic and phototrophic technologies using unbiased metrics (i.e., COD removal for anaerobic technologies, nutrient removal for phototrophic technologies, and potential energy production/consumption for both). As the number of technologies continues to increase, it becomes imperative to determine which have advantageous characteristics and how best to leverage them. By promoting technologies that have synergies between effluent quality and energy production, academia and industry are better poised to advance sustainable practices when beginning a new project. In addition, this work details the benefits of a potential combined anaerobic-phototrophic wastewater treatment plant that would both reduce energy demand and improve energy production. This work does acknowledge that a great deal of research is needed in order for a design such as this to be feasible as a full-scale plant.

## 4.3 Accelerating Sustainability through Quantitative Sustainable Design

This thesis is also novel in its use of a QSD framework that implemented LCA and LCC to elucidate the relative importance and tradeoffs stemming from several design decisions that need to be made when designing an AnMBR. Uncertainty and sensitivity analyses were also utilized to ascertain which decisions and inputs have the greatest effect on net present cost as well as environmental impacts of the design. Certain choices result in a plant that is prohibited by cost, environmental effects, or both (e.g., gas sparging). By identifying these critical barriers, future research can focus on reducing these negative impacts or leveraging the choices that have more beneficial results. This work has also determined that certain designs can actually reduce environmental impacts during operation (i.e., decrease $CO_2$, $NO_x$, and $H^+$ concentrations in environment), which when combined with the findings of Smith et al., 2014 that AnMBRs can be energy positive [224], is a compelling argument for striving to improve the efficiency of AnMBR processes. This framework, while tailored to AnMBRs, can be modified to evaluate any technology landscape to compare potential costs and environmental impacts against present and future benefits.

The contributions of this work extend beyond academia; wastewater practitioners will also be able to better inform the decision-making process by giving insight into which aspects of design influence cost and environmental impact the most. Engineers in industry need to be confident that these alternative wastewater treatment processes will be able to meet discharge requirements in addition to being more efficient than conventional aerobic processes. Ultimately, this research may help transition the state of the practice away from energy- and environment-intensive aerobic wastewater treatment plants while continuing to accomplish the overarching goal of holding human and environmental health paramount.

# CHAPTER 5

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# APPENDIX A

# SUPPORTING INFORMATION FOR ENERGY POSITIVE DOMESTIC WASTEWATER TREATMENT: THE ROLES OF ANAEROBIC AND PHOTOTROPHIC TECHNOLOGIES

## A.1 Methodology for Inclusion in Review and Results

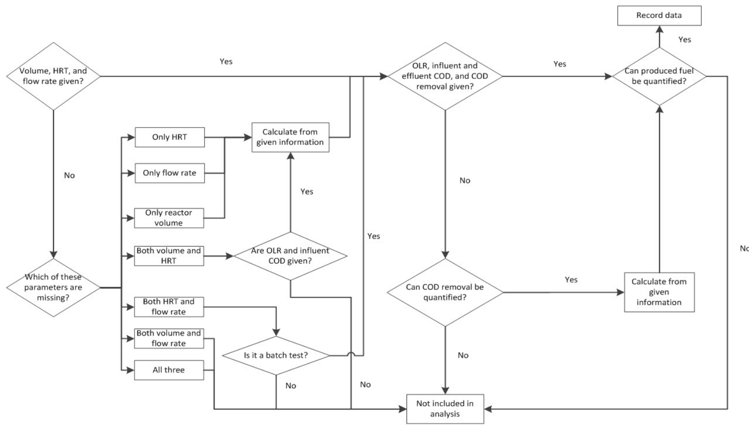


Figure A.1: Decision tree for anaerobic data. This methodology was followed to determine if a manuscript examining an anaerobic technology had sufficient data to be included in the review.

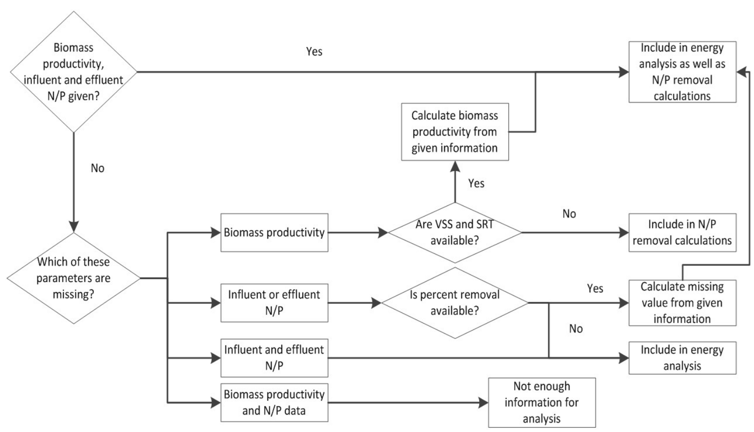


Figure A.2: Decision tree for phototrophic data. This methodology was followed to determine if a manuscript examining a phototrophic technology had sufficient data to be included in the review.

Table A.1: Technologies examined in this review for both anaerobic and phototrophic systems, and the number of published studies included and the respective citations for each.

| **Technology** | **Number of Published Studies** | **Citations** |
|---|---|---|
| ASBR | 4 | [80, 225–227] |
| UASB | 4 | [228–231] |
| ABR | 3 | [232–234] |
| AFB | 5 | [28, 235–238] |
| AnMBR | 6 | [38, 41, 42, 239–241] |
| MEC | 7 | [60, 96–100, 242] |
| MFC | 4 | [60, 95, 243, 244] |
| HRAP | 7 | [115, 245–250] |
| PBR | 2 | [250, 251] |
| Stirred Tank | 8 | [123, 124, 174, 252–256] |
| WSP | 5 | [67, 138, 257–259] |
| ATS | 2 | [145, 146] |

## A.2 Energy Normalization

### A.2.1 Energy Normalization Equations

$$\left(\frac{\text{X mol CH}_4}{}\right)\left(\frac{803\text{kJ}}{1\text{ mol CH}_4}\right) \tag{A.1}$$

$$\left(\frac{\text{X mol H}_2}{}\right)\left(\frac{286\text{kJ}}{1\text{ mol H}_2}\right) \tag{A.2}$$

$$\left(\frac{\text{X kWh}}{}\right)\left(\frac{3600\text{kJ}}{1\text{ kWh}}\right) \tag{A.3}$$

### A.2.2 Assumptions for Anaerobic and Phototrophic Energy Normalization/ Conversion to Usable Energy

Table A.2: Key assumptions utilized when converting from output fuel (anaerobic) or produced biomass (phototrophic) to energy.

| Parameter | Value | Citation |
|---|---|---|
| Energetic content of methane | 803 kJ$\cdot$mol$^{-1}$ | [26] |
| Energetic content of hydrogen gas | 286 kJ$\cdot$mol$^{-1}$ | [260] |
| Gas conversion efficiency (using fuel cell) | 42.30% | [158] |
| N/P limited determination (Redfield ratio) | <16 is N limited | [111] |
| Low COD/VSS ratio | 1.47 g COD$\cdot$g VSS$^{-1}$ 10/40/50% Lipids/Carbs/Proteins | [111, 125, 151–153] |
| High COD/VSS ratio | 1.84 g COD$\cdot$g VSS$^{-1}$ 30/20/50% Lipids/Carbs/Proteins | [111, 151–153] |
| Lipid storage compound | Stearic acid ($C_{18}H_{36}O_2$) | [261] |
| Carbohydrate storage compound | Glucose ($C_6H_{12}O_6$) | [261] |
| Protein compound | $C_{16}H_{24}O_5N_4$ | [154] |
| Energy content of COD | 3.86 kWh$\cdot$kg COD$^{-1}$ | [55] |
| Low biocrude yield | 25% | [262] |
| High biocrude yield | 54% | [263] |
| Higher heating value for microalgae (HTL) | 33.2 MJ$\cdot$kg$^{-1}$ | [164] |
| Lipid content of algal cells | 10-30% | [67] |
| Energy content of biodiesel | 37.2 MJ$\cdot$kg$^{-1}$ | [67] |
| Low methane production from algae digestion | 0.1 L $CH_4$$\cdot$g VSS$^{-1}$ | [264] |
| High methane production from algae digestion | 0.49 L $CH_4$$\cdot$g VSS$^{-1}$ | [160] |
| Combustion energy yield | 14.2-21.4 MJ$\cdot$kg$^{-1}$ | [178, 179] |
| Combustion efficiency | 70% | [67] |
| COD production$\cdot$capita$^{-1}$$\cdot$d$^{-1}$ | 180 | [166] |
| N production$\cdot$capita$^{-1}$$\cdot$d$^{-1}$ | 13 | [166] |
| P production$\cdot$capita$^{-1}$$\cdot$d$^{-1}$ | 2.1 | [166] |

### A.2.3 Energy Normalization Results

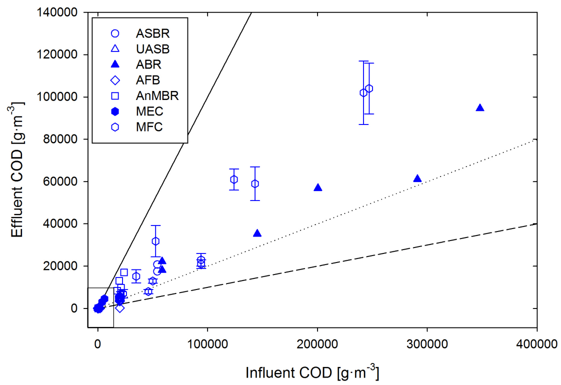


Figure A.3: Influent vs. effluent COD [g·m$^{-3}$] for each paper focusing on anaerobic treatment. Marker shape indicates technology type (e.g. suspended growth, sludge blanket, etc.). The solid line is no COD removal, the dotted line is 80% removal, and the dashed line is 90% removal. The box near the origin is the area examined in Figure 2.5 signifying influent CODs around 500 mg·L$^{-1}$.

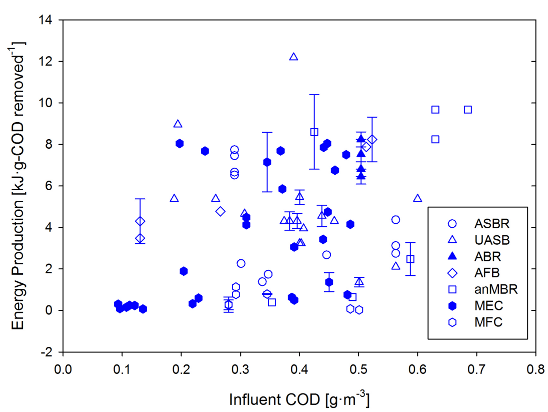


Figure A.4: Energy production [kJ·g-COD removed$^{-1}$] vs. influent COD [g·m$^{-3}$] for each paper studying anaerobic technologies with an influent COD below 500 mg·L$^{-1}$ (for synthetic wastewater) or using actual domestic wastewater.

## A.3 Energy Production

### A.3.1 Theoretical Maximum Energy Production for Anaerobic Technologies

Theoretical maximum energy from anaerobic processes was calculated first by determining the moles of electrons present in the oxidation of 1 kg of oxygen gas to water (see below) [25]

$$\frac{1}{4}\mathrm{O_2} + \mathrm{H^+} + \mathrm{e^-} \rightarrow \frac{1}{2}\mathrm{H_2O}$$
$$\left(\frac{\mathrm{1kg\ COD}}{}\right)\left(\frac{10^3\mathrm{g}}{\mathrm{1kg}}\right)\left(\frac{\mathrm{1mol\ O_2}}{\mathrm{32g\ O_2}}\right)\left(\frac{\mathrm{4mol\ e^-}}{\mathrm{1mol\ O_2}}\right) = \frac{\mathrm{125mol\ e^-}}{\mathrm{1kg\ COD}} \tag{A.4}$$

Once this was determined, the 125 moles of electrons were then converted to energy using thermodynamic half reactions [25] as follows:

Methane:

$$\frac{1}{8}\mathrm{CO_2}+\mathrm{H^+} + \mathrm{e^-} \rightarrow \frac{1}{8}\mathrm{CH_4} + \frac{1}{4}\mathrm{H_2O}$$
$$\left(\frac{\mathrm{125mol\ e^-}}{\mathrm{1kg\ COD}}\right)\left(\frac{\mathrm{1mol\ CH_4}}{\mathrm{8mol\ e^-}}\right)\left(\frac{\mathrm{22.4L\ CH_4}}{\mathrm{1mol\ CH_4}}\right)\left(\frac{35,845\mathrm{kJ}}{\mathrm{1000L\ CH_4}}\right) = \frac{12,500\mathrm{kJ}}{\mathrm{1kg\ COD}} \tag{A.5}$$

Hydrogen gas:

$$\mathrm{H^+} + \mathrm{e^-} \rightarrow \frac{1}{2}\mathrm{H_2}$$
$$\left(\frac{\mathrm{125mol\ e^-}}{\mathrm{1kg\ COD}}\right)\left(\frac{\mathrm{1mol\ H_2}}{\mathrm{2mol\ e^-}}\right)\left(\frac{\mathrm{286kJ}}{\mathrm{1mol\ H_2}}\right) = \frac{17,800\mathrm{kJ}}{\mathrm{1kg\ COD}} \tag{A.6}$$

Electricity:

$$\mathrm{dG} = -\mathrm{emf} \cdot \mathrm{n} \cdot \mathrm{F}$$
$$\mathrm{dG} = -0.69\mathrm{V}\left(\frac{\mathrm{125mol\ e^-}}{\mathrm{1kg\ COD}}\right)\left(\frac{96,485.3\mathrm{C}}{\mathrm{1mol\ e^-}}\right)\left(\frac{\mathrm{1kJ}}{10^3\mathrm{J}}\right) = \frac{8,300\mathrm{kJ}}{\mathrm{1kg\ COD}} \tag{A.7}$$

where dG is the Gibbs free energy, emf is the electromotive force of a microbial fuel cell at open circuit (0.69 V, [92]), n is moles of $e^-$, and F is Faraday's constant (96,485.3 C·mol $e^{-1}$).

### A.3.2 Energy Production for Phototrophic Processes

The average energy production per g-N removed by each technology was as follows (average standard deviation; largest to smallest): PBR (760 ± 250 kJ·g-$N^{-1}$), ATS (300 ± 160 kJ·g-$N^{-1}$), and HRAP (210 ± 96 kJ·g-$N^{-1}$). The average energy production per g-P removed was as follows: HRAP (2,000 ± 1,300 kJ·g-$P^{-1}$), Stirred Tank (2,500 ± 1,600 kJ·g-$P^{-1}$), ATS (1,600 ± 780 kJ·g-$P^{-1}$), and PBR (640 ± 180 kJ·g-$P^{-1}$).

## A.4 Energy Consumption

### A.4.1 Mechanical Mixing

$$G = \sqrt{\frac{P}{\mu_w V_r}} \tag{A.8}$$

where G is the mixing intensity, w is the viscosity of wastewater (N·s·$m^{-2}$), P is power (W), and $V_r$ is volume of the reactor ($m^3$).

### A.4.2 Gas Sparging

$$P_w = \frac{wrT_1}{29.7ne}\left[\left(\frac{p_2}{p_1}^{0.283}\right) - 1\right] \tag{A.9}$$

where $P_w$ is the power requirement (kW), w is the weight flow rate of air - volumetric flow rate of air, $Q_a$, times specific weight - (kg·$s^{-1}$), R is the engineering gas constant for air (8.314 kJ·$kmol^{-1}$·$K^{-1}$, $T_1$ is the absolute inlet pressure (K), $p_1$ is the absolute inlet pressure (atm), $p_2$ is the absolute outlet pressure (atm), n is 0.283, and e is the efficiency (0.80) [26].

For phototrophic systems, the flow rate $Q_a$ necessary to obtain well-mixed algal cultures is assumed to be between 0.1 and 0.3 $L_{air}$· $L_{reactor}^{-1}$·$min^{-1}$ [104, 176].

### A.4.3 Paddlewheel Operation

We assume 0.037 kW·paddle wheel$^{-1}$ and 11,668 MJ·hectare$^{-1}$·year, with 10 paddles per hectare [265].

### A.4.4 Harvesting

Harvesting was calculated using reported energy values from Sturm and Lamer 2011 [67] that included comparisons to Lardon et al. [72], Batan et al. [71], and Stephenson et al. [118]. In all cases, the energy consumption was greater for Sturm and Lamer, and these have been chosen as conservative estimates for coagulation/flocculation, dewatering (belt filter press), and centrifugation. A low estimate was chosen as a combination of coagulation/flocculation and belt filter press for dewatering, while the high range was centrifugation (assuming only gravity settling beforehand). Values from Sturm and Lamer are as follows:

Coagulation/flocculation (1 HP/2,500 gpm skimmer; 70% efficiency (FRC Systems International, Cumming, GA)): 84 kWh·d$^{-1}$

Dewatering (Based on a 250-cm belt width (Komline-Sanderson Model GRS-1)): 340 kWh·d$^{-1}$

Centrifugation: 2,080 kWh·d$^{-1}$

Normalizing these values for the theoretical plant in Sturm and Lamer (12 MGD treated) and converting to Joules, these values become:

Coagulation/flocculation: 6.66 kJ·m$^{-3}$

Dewatering: 26.9 kJ·m$^{-3}$

Centrifugation: 165 kJ·m$^{-3}$

### A.4.5 Pumping (Draw-down, Permeate, and Recirculation)

The total dynamic head (TDH) is comprised of the static head ($H_{ts}$), the friction head ($H_{sf}$, $H_{df}$), and minor losses ($H_m$). For AnMBRs, permeate pumping must also be considered by including transmembrane pressure (TMP). TDH can then be calculated by the equation below:

$$TDH = H_{ts} + H_{sf} + H_{df} + H_m(+TMP) \tag{A.10}$$

However, because minor losses are insignificant compared to the static and friction heads, $H_m$ can be negated. Terms in the TDH equation are further elaborated below:

Total Static Head, $H_{ts}$ (ft): The total static head of pumping can be calculated by the equation below.

$$H_{ts} = H_{ds} - H_{ss} \tag{A.11}$$

Suction Static Head, $H_{ss}$ (ft): Suction static head of pumping is the elevation difference between the water level in the reactor and the centerline of the permeate pump.

Discharge Static Head, $H_{ds}$ (ft): Discharge static head of pumping is the elevation difference between the centerline of the pump and the centerline of the effluent (where water is discharged). The effluent is assumed to be the highest point, thus setting the hydraulic reference.

Suction Friction Head, $H_{sf}$ (ft): Suction friction head can be estimated using the Hazen-Williams equation. Suction friction head refers to the friction loss caused in the pipes on the suction side.

$$H_{sf} = 3.02LV^{1.85}C^{-1.85}D^{-1.17} \tag{A.12}$$

where L is the length of the pipe (ft), V is the velocity of the liquid in the pipe ($ft{\cdot}s^{-1}$), D is the inner diameter of the pipe (ft) and C is the Hazen-Williams coefficient (110).

Discharge Friction Head, $H_{df}$ (ft): Discharge friction head refers to the friction loss caused in the pipes on the discharge side. The Hazen-Williams equation is also used to calculate this value.

Transmembrane Pressure, TMP (ft): This value is based on is based on the 25th and 75th percentiles of TMP reported in the literature [33, 34, 42, 44, 240, 241, 266–268].

Brake Horsepower, BHP: BHP is the amount of horsepower required to drive the pump and can be calculated by the equation below:

$$BHP = \frac{Q \cdot TDH}{3,960 \cdot \text{Pump Efficiency}} \tag{A.13}$$

where Q is the flow rate (gpm), TDH is the total dynamic head (ft), and the pump efficiency is assumed to be 80%.

Energy consumption, E (kW): The amount of energy input into the motor of the pump can be calculated as:

$$E = \frac{0.746 \cdot BHP}{\text{Motor Efficiency}} \tag{A.14}$$

where BHP is the break horsepower (hp) and motor efficiency is assumed to be 70%.

### A.4.6 Heating

We can compute the heat requirement for the influent wastewater stream as follows [26]:

$$q = \frac{cm\Delta T}{Q} \tag{A.15}$$

where q is heat transfer ($kJ \cdot m^{-3}$), c is the specific heat of the wastewater stream ($J \cdot kg^{-1} \cdot {}^{\circ}C^{-1}$), m is mass flow rate of wastewater ($kg \cdot d^{-1}$), Q is the flow rate ($m^3 \cdot d^{-1}$) and $\Delta$T is the temperature change (°C).

### A.4.7 Assumptions for Energy Consumption Calculations

Table A.3: Assumptions associated with each of the energy-consuming processes listed above along with values and citations thereof.

| **Assumption** | **Value** | **Citation** |
|---|---|---|
| **General** | | |
| Influent pumping | 0 $m^3 \cdot d^{-1}$ | - |
| HRT | 1 day | - |
| Influent flow rate | 1,000 $m^3 \cdot d^{-1}$ | - |
| Influent COD | 500 $mg \cdot L^{-1}$ | [55] |
| Influent N | 36 $mg \cdot L^{-1}$ | [55] |
| Influent P | 5.8 $mg \cdot L^{-1}$ | [55] |
| Specific heat of wastewater | 4.2 $kJ \cdot kg^{-1} \cdot {}^\circ C^{-1}$ | [26] |
| Density of wastewater (20C) | 998.2063 $kg \cdot m^{-3}$ | [84] |
| Viscosity of water | $8.9 \text{x} 10^{-4}$ $N \cdot s \cdot m^{-2}$ | [84] |
| Specific weight of water (20C) | 1.20 $kg \cdot m^{-3}$ | [26] |
| Pipe internal diameter | 1 ft | - |
| **Mixing** | | |
| Mixing intensity, ASBR | 250-350 $s^{-1}$ | - |
| Mixing intensity, Stirred Tank | 100-200 $s^{-1}$ | - |
| **Gas Sparging** | | |
| Weight flow of air | 13-19 $kg \cdot s^{-1}$ | [33, 34] |
| Temperature | 20 °C | - |
| Outlet pressure | 1.4 atm | - |
| Efficiency | 80% | [26] |
| **Aeration** | | |
| Weight flow of air | 2-4 $kg \cdot s^{-1}$ | - |
| Temperature | 20°C | - |
| Outlet pressure | 1.4 atm | - |
| Efficiency | 80% | [26] |
| **Paddlewheel Operation** | | |
| Number of paddlewheels | 1-3 | - |
| Area of reactor | 3,030.3 $m^2$ | - |
| Depth | 0.33 m | - |

Table A.3 (cont.)

| Assumption | Value | Citation |
|---|---|---|
| **Recirculation Pumping** | | |
| Total static head | 2 ft | - |
| Suction friction head length | 30 ft | - |
| Discharge friction head length | 120 ft | - |
| Flow rate for BHP calculation | 5x Influent flow | - |
| Velocity in pipes | 3-6 ft·s$^{-1}$ | - |
| Recirculation Rate | 4.4-386.6 | [236] |
| **Effluent Pumping** | | |
| Total static head | 5 ft | - |
| Suction friction head length | 30 ft | - |
| Discharge friction head length | 30 ft | - |
| Flow rate for BHP calculation | Influent flow | - |
| Velocity in pipes | 3-6 fts-1 | - |
| **Permeate Pumping** | | |
| Total static head | 2 ft | - |
| Suction friction head length | 30 ft | - |
| Discharge friction head length | 30 ft | - |
| Flow rate for BHP calculation | Influent flow | - |
| Velocity in pipes | 3-8 ft·s$^{-1}$ | - |
| **Heating**[a] | | |
| Influent temperature | 18-23°C | - |
| Desired heating temperature | 35°C | - |
| Heat transfer efficiency | 100% | - |
| **Applied Voltage** | | |
| Applied Voltage | 0.5-1.2 V | [99] |

[a]For the anaerobic processes, heating occurs before the influent flow

### A.4.8 Conversion Efficiencies for Biomass to Fuel Processes

Table A.4: Energy yield capabilities of biomass conversion processes indicating resulting fuel.

| **Conversion** | **Fuel Type** | **Energy Content [MJ·kg fuel$^{-1}$]** | **Conversion Efficiency [% Biomass Converted to Fuel]** | **Low Energy Yield [MJ]** | **High Energy Yield [MJ]** | **Citation** |
|---|---|---|---|---|---|---|
| Caloric Content | Biomass | 20.4-25.6[a] | 100% | 20.4 | 25.6 | Calculated |
| Hydrothermal Liquefaction | Crude Oil | 33.2 | 25-54% | 8.3 | 17.9 | [164] |
| Anaerobic Digestion | Methane | 35.8 | 40-60%[b] | 3.59 | 17.6 | [26, 160, 185] |
| Transesterification | Biodiesel | 37.2 | 10-30%[c] | 3.72 | 11.16 | [67, 164] |
| Combustion | Biomass | 14.2-21.4 | 70% | 9.9 | 15 | [178, 179] |

[a]Calculated from COD content assuming low and high composition (10 and 30% lipid) as listed in the manuscript and a theoretical 13.9 kJ·g COD$^{-1}$ oxidized to $CO_2$ and $H_2O$ [157]
[b]Estimated conversion efficiency of anaerobic digestion on algal VSS only [160]. For calculations, conversions of 0.1 and 0.49 $m^3$ $CH_4$·kg VSS$^{-1}$ used [160]
[c]Lipid content, assumed 100% conversion of lipids to biodiesel [67, 125]

Table A.5: Energy consumption of technologies shown in Table 2, converted to kJ·g nutrient removed$^{-1}$ for ease of comparison to energy yields in Table 4.

| Technology | E [kJ]·day$^{-1}$ (low) | E [kJ]·day$^{-1}$ (high) | E [kJ]·g N removed$^{-1}$ (low) | E [kJ]·g N removed$^{-1}$ (high) | E [kJ]·g P removed$^{-1}$ (low) | E [kJ]·g P removed$^{-1}$ (high) |
|---|---|---|---|---|---|---|
| HRAP | 37,200 | 179,600 | 2 | 7 | 12 | 59 |
| PBR | 6,389,000 | 13,228,000 | 226 | 468 | 1,181 | 2,446 |
| Stirred Tank | 832,000 | 3,301,000 | 37 | 147 | 183 | 728 |
| WSP | 3,400 | 170,000 | N/A | N/A | N/A | N/A |
| ATS | 0 | 0 | 0 | 0 | 0 | 0 |

## A.5 Treatment system based energy simulation for the Strass, Austria wastewater treatment plant

Table A.6: Simulated energy consumption, recovery and production associated with COD and N removal at the Strass wastewater plant. Refer to Figure A.5 for more detail.

| Condition | Stage B Energy [kJ·Cap$^{-1}$] | | Digester Biogas [kJ·Cap$^{-1}$] | Total [kJ·Cap$^{-1}$] |
|---|---|---|---|---|
| | COD | $NH_3$-N | | |
| BNR | | -140 | 740 | 600 |
| ABR + HRAP | 420 | 2,200 | 600 | 3,200 |
| ABR + PBR | 420 | 7,900 | 600 | 8,900 |

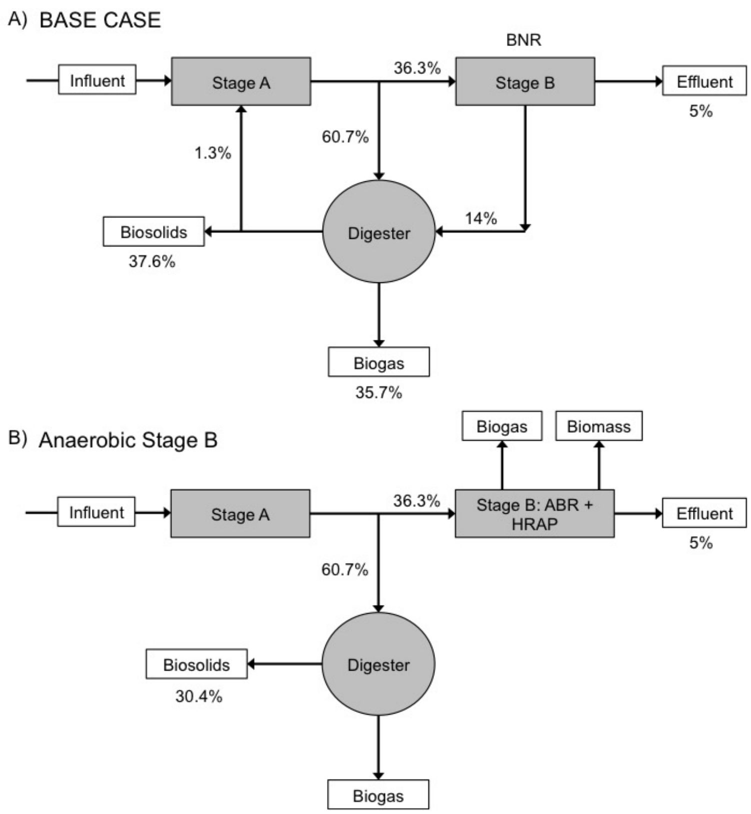


Figure A.5: Process flow diagrams of the Strass wastewater treatment plant. A) Base Case: Currently the plant employs aerobic biological nutrient removal (BNR) to treat the liquid stream. B) Anaerobic Stage B: If the BNR stage was converted to a two stage process involving COD removal in a anaerobic baffled reactor (ABR) and nutrient removal in a high rate algal pond (HRAP), COD and nitrogen removal would result in the production of biogas and biomass. The percentages refer to the COD mass balance within the plant.

# APPENDIX B

# SUPPORTING INFORMATION FOR ADVANCING ANMBR DESIGN BY IDENTIFYING A PATHWAY FOR SUSTAINABLE TECHNOLOGY INNOVATION

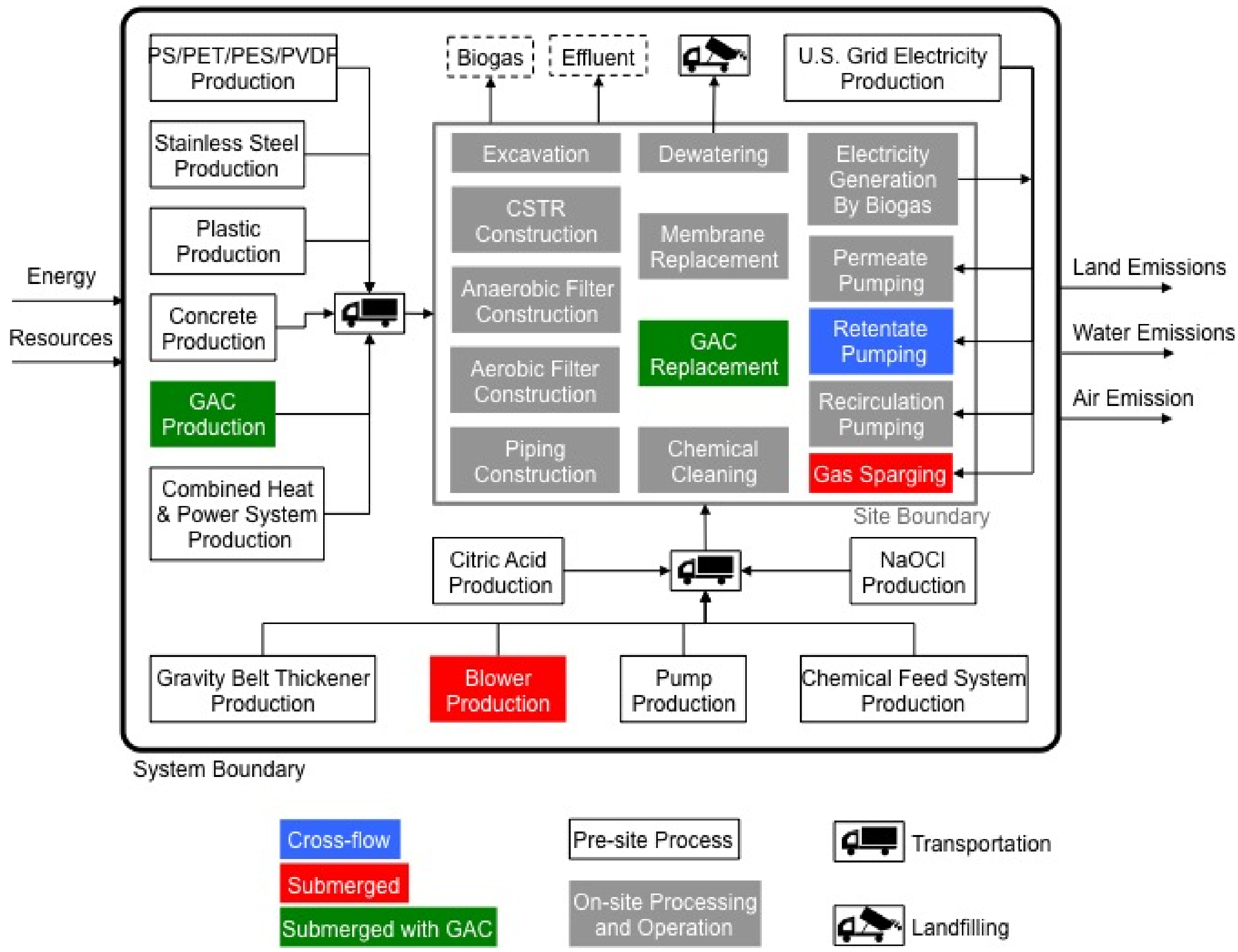


Figure B.1: System boundary for LCA/LCC of AnMBRs, including both construction and operation. First- and second-order environmental impacts are considered. Impacts pertaining to specific designs are indicated.

Table B.1: Key static assumptions utilized when calculating outputs.

| **General** | | |
|---|---|---|
| Greenfield construction | | |
| Effluent standards met by all designs | | |
| GAC not replaced | | |
| Including GAC eliminates need for gas sparging | | |
| DM not replaced | | |
| DM is 100% efficient at removing dissolved methane | | |
| Electricty generation is based on the state of Illinois | | |
| Slope of 1.5:1 and a freedboard of 3 ft were used for excavation | | |
| **Parameter** | **Value** | **Citation** |
| Cost of GAC | $1.50·lb$^{-1}$ | - |
| Cost of Electricity | $0.07·kWh$^{-1}$ | [a] |
| Cost of Degassing Membrane | $10,000·DM$^{-1}$ | [b] |
| Flow Rate through DM | 30 m$^3$·hr$^{-1}$ | [b] |
| Power Consumption by DM | 0.0042 kW | [194] |
| Fuel Cell CHP Efficiency | 40.50% | [158] |
| Microturbine CHP Efficiency | 27% | [221] |
| Internal Combustion CHP Efficiency | 36% | [158] |
| Combustion Gas CHP Efficiency | 31.50% | [158] |
| Membrane useful life | 7 years | [a] |

[a] Hazen and Sawyer, personal communication, April 19, 2013
[b] Christine Medordi, personal communication, June 19, 2014

Table B.2: Parameters varied during Monte Carlo Simulation.

| Parameter | Units | Distribution | Min | Max | Average | St. Dev. |
|---|---|---|---|---|---|---|
| Flux | $L \cdot m^{-2} \cdot hr^{-1}$ | Normal | - | - | 8.5 | 2.5 |
| TMP[a] (high) | psi | Normal | - | - | 25 | 2 |
| TMP (low) | psi | Normal | - | - | 20 | 2 |
| OL[b] (AF) | g-COD$\cdot L^{-1} \cdot d^{-1}$ | Uniform | 4 | 6 | - | - |
| HL[c] (AF) | $m \cdot hr^{-1}$ | Uniform | 8 | 12 | - | - |
| OL (AeF) | g-COD$\cdot L^{-1} \cdot d^{-1}$ | Uniform | 1 | 3 | - | - |
| HL (AeF) | $m \cdot hr^{-1}$ | Uniform | 1 | 3 | - | - |
| HRT[d] | hr | Uniform | 8 | 12 | - | - |
| IRR[e] | Multiple of forward flow | Uniform | 2 | 6 | - | - |
| Interest | % | Uniform | 6 | 10 | - | - |
| GAC Dose | $g \cdot L^{-1}$ | Uniform | 1.7 | 225 | - | - |
| Cross-flow Velocity | $m \cdot s^{-1}$ | Uniform | 1 | 5 | - | - |
| SGD[f] | $m^3 \cdot m^{-1} \cdot hr^{-1}$ | Uniform | 0.4 | 3 | - | - |
| Sparging frequency | % | Uniform | 50 | 100 | - | - |
| SRT[g] | d | Uniform | 20 | 50 | - | - |
| Percent gaseous methane | % | Uniform | 50 | 70 | - | - |
| Flow Rate | MGD | Normal | - | - | 20 | 5 |
| SS0[h] | mg-COD$\cdot L^{-1}$ | Normal | - | - | 300 | 30 |
| XS0[i] | mg-COD$\cdot L^{-1}$ | Normal | - | - | 100 | 10 |

[a]Transmembrane pressure.
[b]Organic loading rate.
[c]Hydraulic loading rate.
[d]Hydraulic retention time.
[e]Internal recirculation rate.
[f]Specific gas demand.
[g]Solids retention time.
[h]Soluble influent substrate.
[i]Particulate influent substrate.

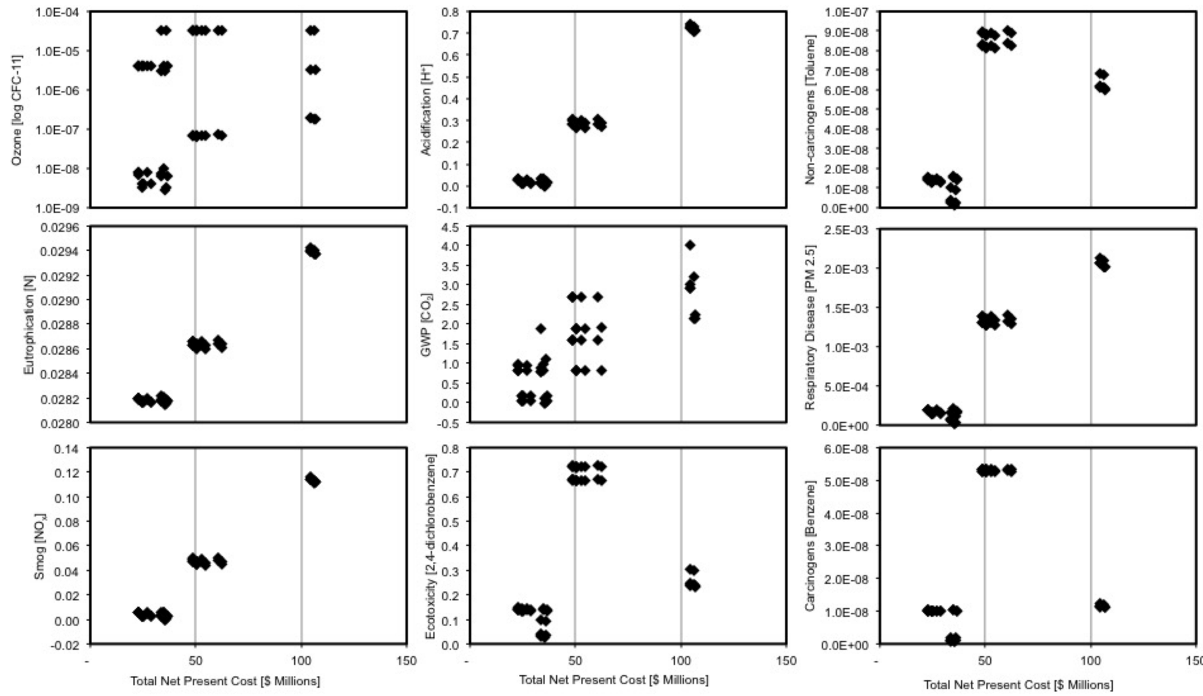


Figure B.2: Total net present life cycle costs [$ Million] compared to the nine environmental impacts examined. All environmental impacts are kg equivalents per $m^3$ treated.

Table B.3: Equations and ranges from CAPDETWorks were edited in order to make them continuous. Alterations in the equation or the applicable range are bolded and italicized. Installed equipment cost for pumping was replaced by a polynomial curve fit to the output of the original equation.

| **Pumping** | |
|---|---|
| **Firm Pumping Capacity Range** | |
| 7-***41*** | Equation unchanged |
| ***41***-80 | Equation unchanged |
| **Installed equipment cost** | |
| Curve fit to data | $\boldsymbol{2.065 \cdot 10^5 + 7.721 \cdot 10^4 \cdot Q_{avg}}$ |

[a]Capacity of individual blower

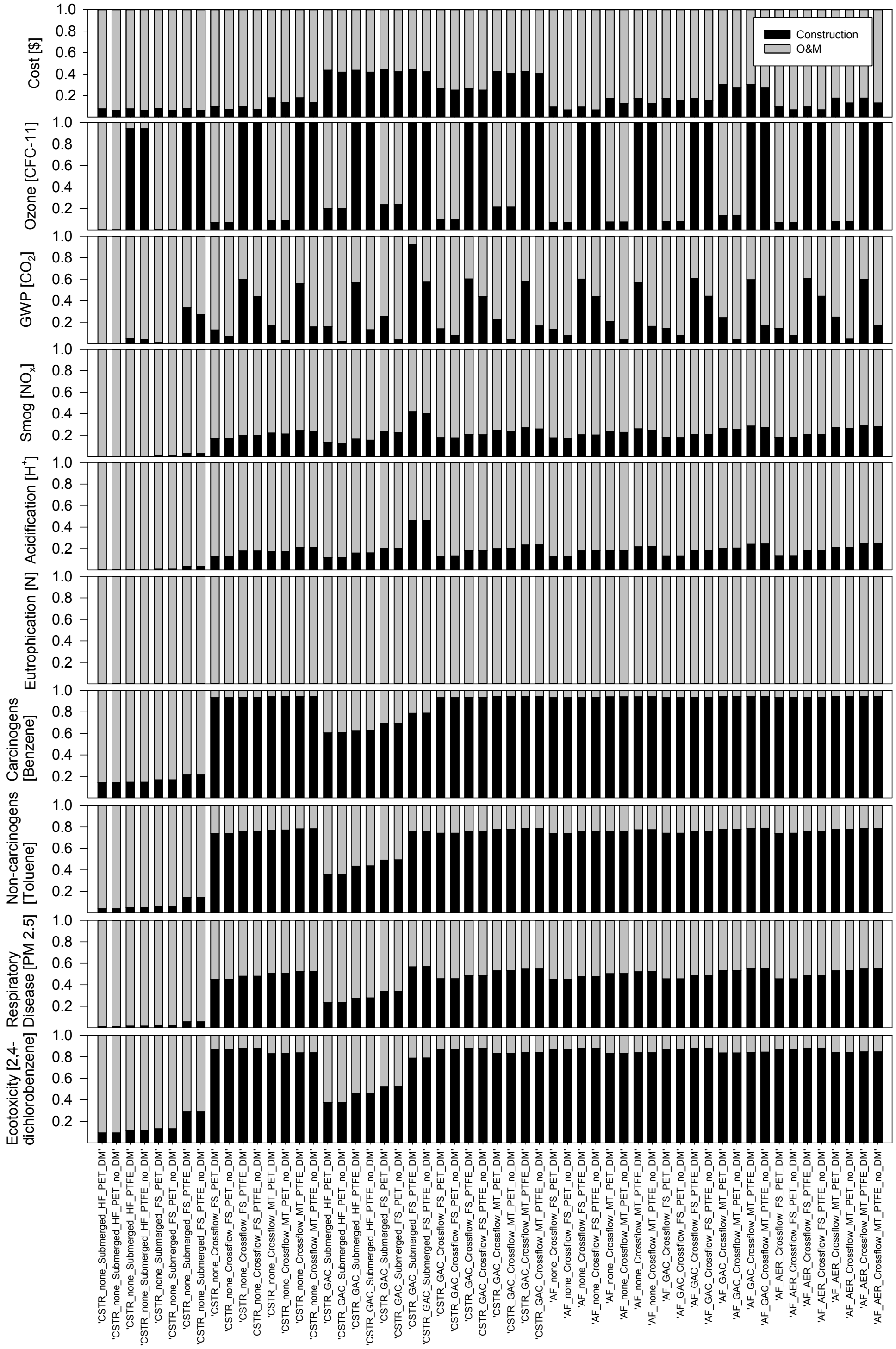


Figure B.3: Percent contribution of construction and O&M to net present life cycle cost and environmental impacts for the 56 designs considered (i.e., after excluding CHP choices).

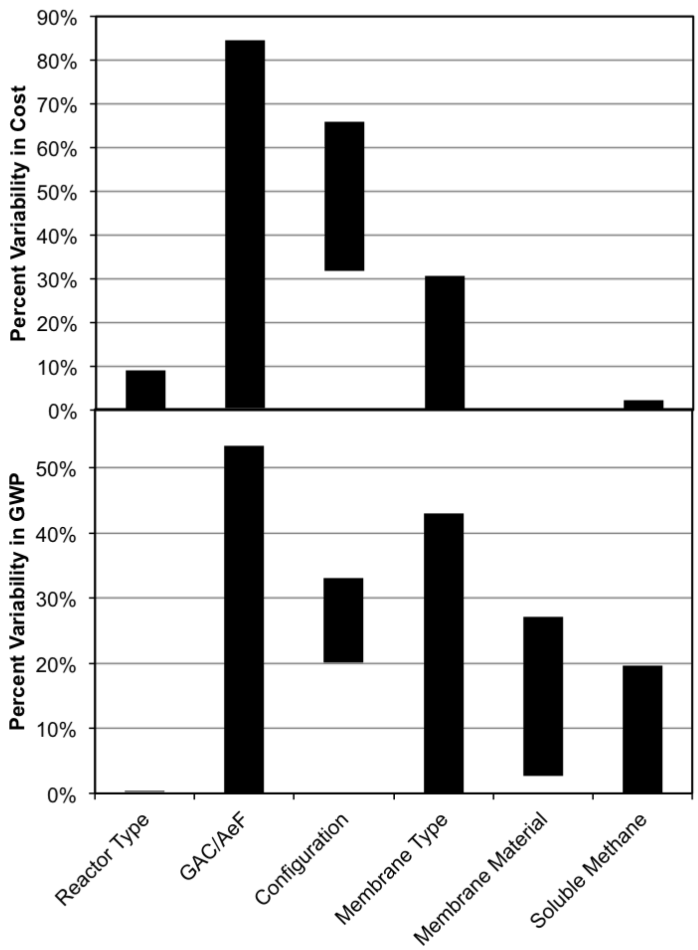


Figure B.4: Percent change in cost and GWP resulting from the six design decisions outlined in Figure 3.1. Values were calculated by normalizing the maximum and minimum difference between two configurations that only varied by one decision to the overall range (i.e., across all 56 configurations).

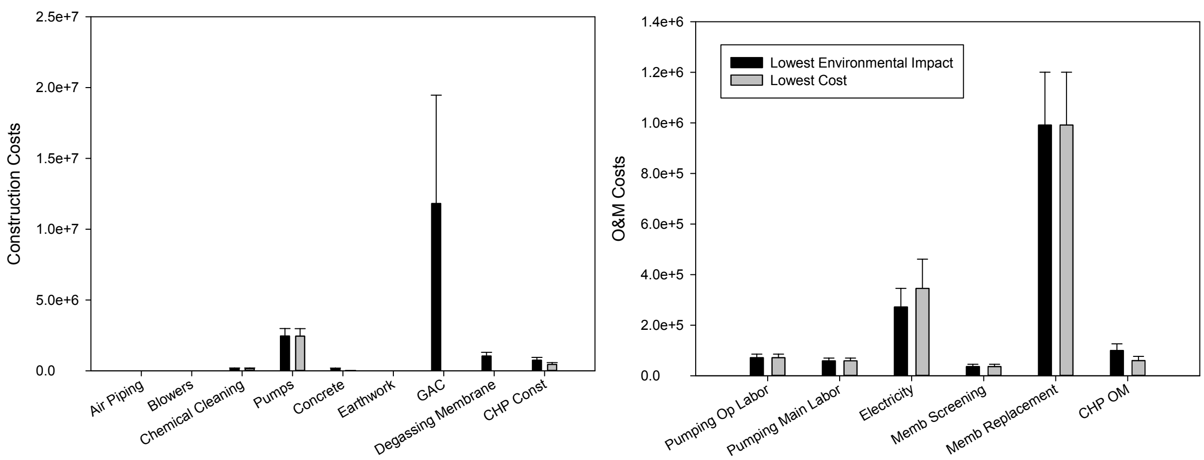


Figure B.5: Breakdown of contributors to construction and O&M costs for the design that had the lowest cost and the design with the lowest environmental impacts.

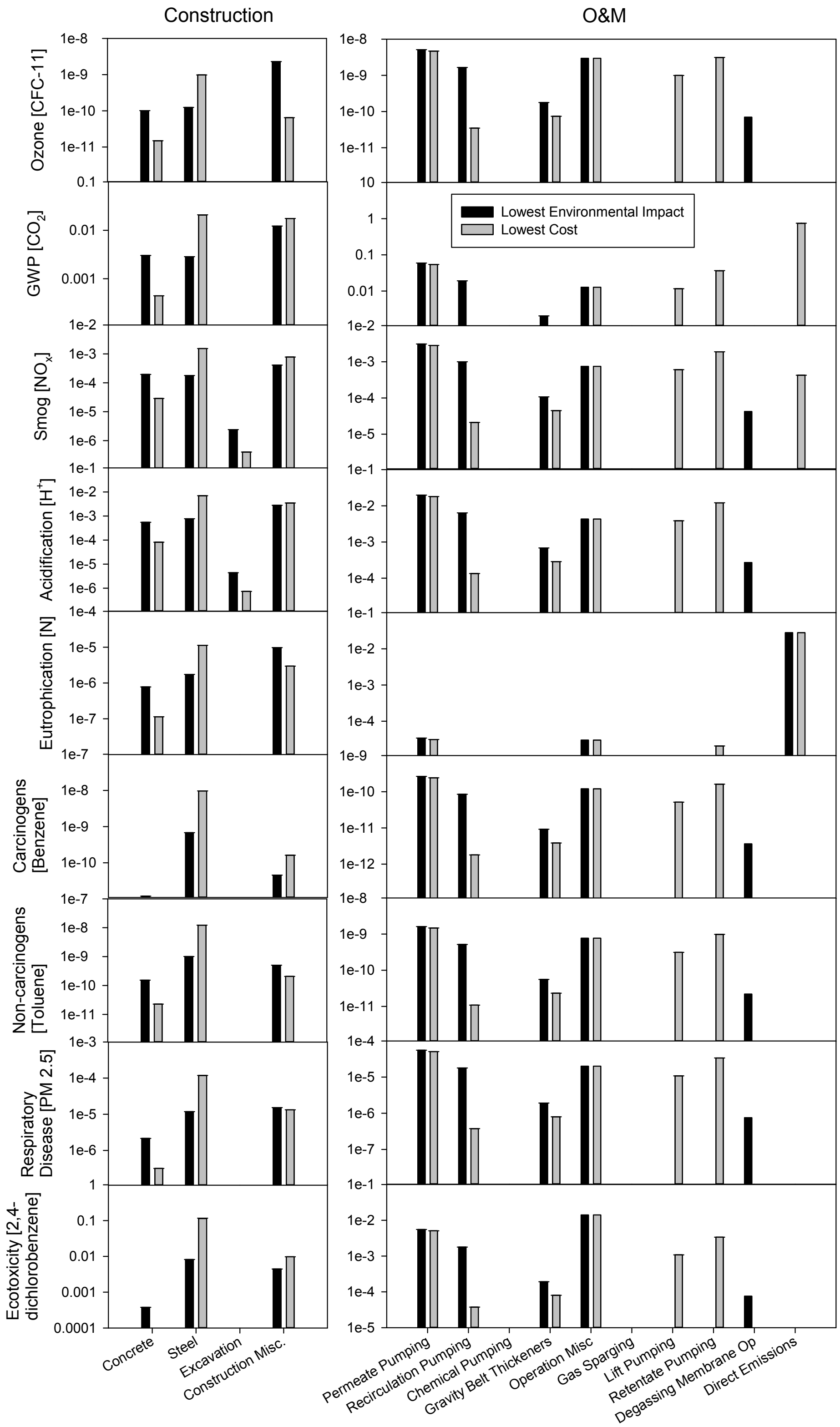


Figure B.6: Breakdown of contributors to construction and O&M environmental impacts (for all nine impacts examined) for the design that had the lowest cost and the design with the lowest environmental impacts.

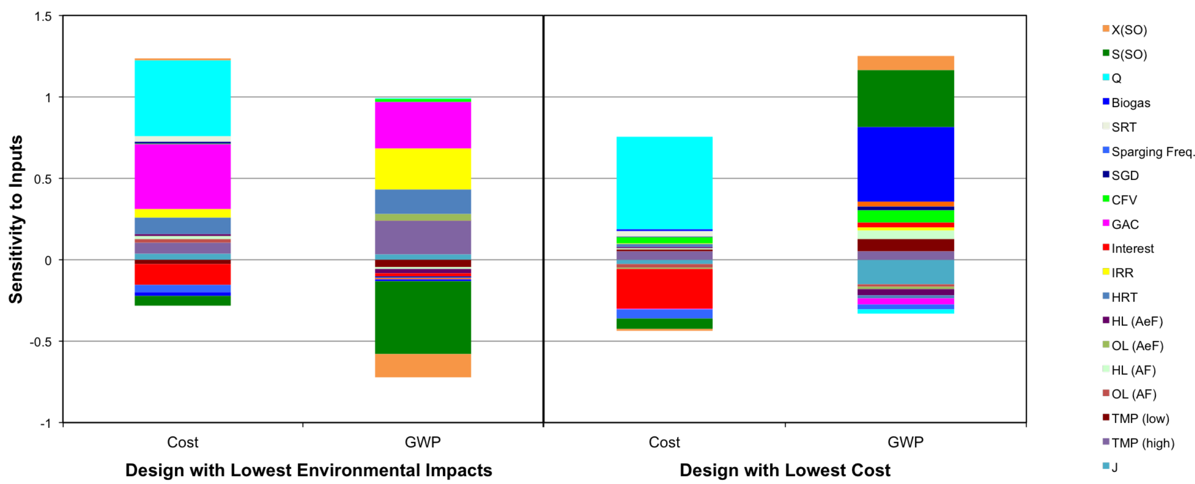


Figure B.7: Sensitivity to inputs for total net present life cycle cost [$] and GWP [kg $CO_2$ eq$\cdot$m$^3$treated$^{-1}$ for the configurations with the lowest total cost or environmental impacts. Sensitivity was determined using Spearman's rank correlation.